JYU DISSERTATIONS 226

Eini Nieminen

How to Protect Nature – Boreal Mire Conservation in Finland

UNIVERSITY OF JYVÄSKYLÄ
FACULTY OF MATHEMATICS AND SCIENCE

JYU DISSERTATIONS 226

Eini Nieminen

How to Protect Nature – Boreal Mire Conservation in Finland

Esitetään Jyväskylän yliopiston matemaattis-luonnontieteellisen tiedekunnan suostumuksella
julkisesti tarkastettavaksi kesäkuun 5. päivänä 2020 kello 12.

Academic dissertation to be publicly discussed, by permission of
the Faculty of Mathematics and Science of the University of Jyväskylä,
on June 5, 2020 at 12 o'clock noon.

JYVÄSKYLÄN YLIOPISTO
UNIVERSITY OF JYVÄSKYLÄ

JYVÄSKYLÄ 2020

Editors
Jari Haimi
Department of Biological and Environmental Science, University of Jyväskylä
Ville Korkiakangas
Open Science Centre, University of Jyväskylä

Cover photo by Eini Nieminen.

Permanent link to this publication: http://urn.fi/URN:ISBN:978-951-39-8177-8

ISBN 978-951-39-8177-8 (PDF)
URN:ISBN:978-951-39-8177-8
ISSN 2489-9003

"The relationship between people and nature, particularly in the context of protected areas, is highly political, embracing issues of rights and access to land and resources, the role of the state (---), and the power of scientific and other understandings of nature."

Adams and Hutton 2007, page 151.

ABSTRACT

Nieminen, Eini
How to protect nature – Boreal mire conservation in Finland
Jyväskylä: University of Jyväskylä, 2020, 79 p.
(JYU Dissertations
ISSN 2489-9003; 226)
ISBN 978-951-39-8177-8 (PDF)
Yhteenveto: Miten suojella luontoa – Tapauksena soiden suojelu Suomessa
Diss.

Nature's resources enable the existence of humankind. Due to our very intensive resource extraction, many ecosystem functions are deteriorating and species' populations, distributions, and assemblages changing, which jeopardize our contemporary societies. Conservation areas effectively slow down these trajectories. However, conservation benefits and costs are often incommensurable, and spatially and temporally unevenly distributed, so their reliable evaluation is complex. I studied how conservation decision-making could produce ecologically, socially, and economically effective and acceptable conservation solutions. I implemented the study in the context of boreal mire conservation network complementation in Finland. Mires need to be set aside as spatially and functionally continuous entities to safeguard their hydrology and long-term existence. In Finland, mires are mostly privately owned and landownership is fragmented within single mires. As a result, conflicts could not be avoided. However, trade-offs between ecological gains, landowners' conservation preferences, and conservation costs could have been alleviated, if alternative conservation solutions were recognized, their consequences studied, and some of the current legislation revised. Furthermore, not all assumed conflicts proved to be true since landowners of wooded mires did not engage in systematic pre-emptive loggings. The results also show that spatial prioritization methods can fill the science-practice gap by supporting conservation planning and decision-making in diverse ways. They can serve simultaneously as site selection tools and as platforms to decision-making, enhancing sharing and analytical use of expert knowledge. They also allow quantification of interrelationships between different conservation-related factors, which enables informing decision-makers about the consequences of alternative conservation solutions. Boreal mire conservation in Finland reflects the very same challenges than nature conservation around the world, so its solutions can help to resolve global conservation problems.

Keywords: Conservation decision-making; involuntary conservation; peatland; pre-emptive behavior; trade-offs; voluntary conservation; Zonation.

Eini Nieminen, University of Jyväskylä, Department of Biological and Environmental Science, P.O. Box 35, FI-40014 University of Jyväskylä, Finland

TIIVISTELMÄ

Nieminen, Eini
Miten suojella luontoa – Tapauksena soiden suojelu Suomessa
Jyväskylä: Jyväskylän yliopisto, 2019, 79 p.
(JYU Dissertations
ISSN 2489-9003; 226)
ISBN 978-951-39-8177-8 (PDF)
Yhteenveto: Miten suojella luontoa – Tapauksena soiden suojelu Suomessa
Diss.

Ihmiskunta on täysin riippuvainen luonnon ekosysteemien toiminnasta. Olemme kuitenkin valjastaneet luonnon niin tehokkaasti omaan käyttöömme, että monet ekosysteemit ovat heikentyneet ja niiden lajisto uhanalaistunut tai kadonnut. Tämä vaarantaa myös ihmisen tulevaisuuden. Luonnonsuojelualueet estävät ja hidastavat tätä kehitystä, mutta usein suojelu on vaikeaa. Sen haittoja ja hyötyjä ei ole helppo arvioida, koska ne eivät jakaudu tasaisesti ihmisten kesken eivätkä ole yhteismitallisia. Tässä väitöskirjassa tutkin, miten luonnonsuojelun päätöksenteko voisi edistää ekologisesti, yhteiskunnallisesti ja taloudellisesti kestävää suojelua. Tutkimustapauksena hyödynsin soidensuojelun täydennysohjelmaa Suomessa. Ohjelma on ollut konfliktien kirjoma. Alun perin se piti toteuttaa luonnonsuojelulakiin perustuvana suojeluohjelmana mahdollistaen maan lunastukset suojelutarkoituksiin, mutta sittemmin se muutettiin maanomistajan suojeluhalukkuuteen perustuvaksi. Osoitin, että suojeltavan monimuotoisuuden määrä ja suojelun taloudelliset kustannukset riippuvat siitä, millä tavalla maanomistajien mielipiteet huomioidaan suojelusuunnittelussa. Havaitsin, että suojelun ekologisten, taloudellisten ja sosiaalisten näkökulmien välisiä ristiriitoja voidaan vähentää. Tämä kuitenkin edellyttää eri suojeluvaihtoehtojen huomaamista ja niiden seurausten ennakkoanalysointia. Osoitin, että suojelun priorisointimenetelmillä voidaan monin eri tavoin edistää suojelupäätöksentekoa. Ne toki soveltuvat perinteiseen suojelualueiden kohdevalintaan, mutta tämän lisäksi ne mahdollistavat eri suojeluvaihtoehtojen seurausten analysoinnin sekä niin sanotun hiljaisen asiantuntijatiedon jakamisen ja analyyttisen käytön. Havaitsin myös, että kaikki täydennysohjelmaan liitetyt konfliktit eivät ole totta, sillä maanomistajat eivät systemaattisesti aavistushakanneet puustoisia soitaan. Soiden suojelu Suomessa heijastelee haasteita, joita luonnonsuojelu kohtaa kaikkialla maailmassa. Näin ollen suomalaisen luonnonsuojelun ratkaisut voivat auttaa selättämään globaaleja suojelun ongelmia.

Avainsanat: Aavistushakkuu; luonnonsuojelupolitiikka; monimuotoisuus; pakkosuojelu; päätöksenteko; vapaaehtoinen luonnonsuojelu; Zonation.

Eini Nieminen, Jyväskylän yliopisto, Bio- ja ympäristötieteiden laitos, PL 35, 40014 Jyväskylän yliopisto

Author's address Eini Nieminen
Department of Biological and Environmental Science
School of Resource Wisdom
P.O. Box 35
FI-40014 University of Jyväskylä
Finland
eini.m.nieminen@jyu.fi

Supervisors Professor Janne Kotiaho
Department of Biological and Environmental Science
School of Resource Wisdom
P.O. Box 35
FI-40014 University of Jyväskylä
Finland

Docent Panu Halme
Department of Biological and Environmental Science
School of Resource Wisdom
P.O. Box 35
FI-40014 University of Jyväskylä
Finland

Doctor Santtu Kareksela
Parks and Wildlife Finland
P.O. Box 36
FI-40101 Metsähallitus
Finland

Reviewers Doctor Jonas Geldmann
Center for Macroecology, Evolution and Climate
Universitetsparken 15
2100 København Ø, Bygning 3
Denmark

Research Professor Matti Koivula
Natural Resources Institute Finland (Luke)
Latokartanonkaari 9
FI-00790 Helsinki
Finland

Opponent Doctor Anni Arponen
Helsinki Institute of Sustainability Science (HELSUS)
Viikinkaari 1, Biocentre 3
00790 Helsinki
Finland

CONTENTS

LIST OF ORIGINAL PUBLICATIONS

The thesis is based on the following original papers, which will be referred to in the text by their Roman numerals I–III.

I Santtu Kareksela, Kaisu Aapala, Aulikki Alanen, Tuomas Haapalehto, Janne Kotiaho, Joona Lehtomäki, Niko Leikola, Ninni Mikkonen, Atte Moilanen, Eini Nieminen, Seppo Tuominen & Raimo Virkkala 2019. Combining spatial prioritization and expert knowledge facilitates effectiveness of large-scale mire protection process in Finland. *Biological Conservation* 241: 108324.

II Eini Nieminen, Kalle Salovaara, Panu Halme & Janne Kotiaho. No evidence of systematic pre-emptive loggings after notifying landowners of their lands' conservation potential. Submitted manuscript.

III Eini Nieminen, Santtu Kareksela, Panu Halme & Janne Kotiaho. Quantifying trade-offs between ecological gains, economic costs, and landowners' preferences in boreal peatland protection. Submitted manuscript.

The table below shows the contributions of the authors to the original papers. The order of the authors indicates the amount of contribution, the first author having the largest and the last one the smallest share.

	I	II	III
Planning	SK, KA, AA, JSK, AM, TH, JL, NM, NL	EN, KS, JSK, PH	EN, SK, JSK, PH
Data	NL, NM, SK, KA, AA, TH, RV, ST, EN	EN, KS	EN, SK
Analyses	SK, NM, NL, TH, JL, AM	EN, KS	EN, SK
Writing	SK, KA, AA, TH, JSK, JL, NL, NM, AM, EN, ST, RV	EN, JSK, PH, KS	EN, SK, JSK, PH

EN = Eini Nieminen, PH = Panu Halme, SK = Santtu Kareksela, JSK = Janne S. Kotiaho, KS = Kalle Salovaara, KA = Kaisu Aapala, AA = Aulikki Alanen, AM = Atte Moilanen, TH = Tuomas Haapalehto, JL = Joona Lehtomäki, NM = Ninni Mikkonen, NL = Niko Leikola, RV = Raimo Virkkala, ST = Seppo Tuominen

GLOSSARY

connectivity	"The degree to which the landscape facilitates or impedes movement [of organisms] among resource patches." (Taylor *et al*. 1993, p. 571.)
conservation plan	An extensive plan for practical conservation implementation. Includes a conservation solution, means, schedules, roles of different agents, etc.
conservation planning	An extensive process including reserve site selection, choosing conservation means, defining schedules and roles of different agents, etc.
conservation scenario	Combination of ecological, social, economic, and political factors that have an effect on conservation planning and on a conservation solution.
conservation solution	Combination of sites that are proposed for protection.
decision maker	A stakeholder, a politician, a conservation expert, or any other person that is allowed to make decisions concerning conservation planning.
decision making	Any process where issues such as a conservation solution, means or schedules of protection, relevant stakeholders, etc. are fixed.
mire	An ecosystem "dominated by living peat-forming plants" (Rydin and Jeglum 2006, p. 4). In Finland, peat formation and >50% coverage of typical peatland vegetation are required in order to define a habitat as a mire (Eurola *et al*. 2015).
representation	"The occurrence of species (or other attributes) within a set of selected areas." (Williams 2001.)
reserve site selection	Choosing sites for protection.
peat	"…the remains of plant and animal constituents accumulating under more or less water-saturated conditions owing to incomplete decomposition." (Rydin and Jeglum 2006, p. 3.)
stakeholder	A party that has an interest in protection of certain areas and who can affect or be affected by it (the definition adapted from https://urly.fi/1wHt).

1 INTRODUCTION

1.1 Human societies embedded in nature

1.1.1 What is nature?

The answer depends on a definition and its definer, but a natural scientist would strive to reply from the viewpoint that philosophers call realist ontology: only one truth of nature exists independently of its observer (Moon and Blackman 2014). This is different to a relativist worldview where the truth would depend on how the definer's personal mind is constructed. To find the answer to the question, the natural scientist would make empirical measurements that she/he would quantitatively count and test as an aim to verify and generalize the one truth. In philosophy, this way of creating knowledge is called objectivist epistemology as a separation from the opposite, i.e. subjectivist epistemology where one thinks that comprising knowledge depends on an observer; what is knowledge to one is not necessarily knowledge to the other.

Following the ontological worldview and the epistemologists´ way of creating knowledge, natural scientists have constructed multiple concepts such as biodiversity, ecosystem, species, and climate, which all relate to nature one way or another. The abundance of sub-concepts reveals that defining nature is not an easy task. As a natural scientific phenomenon, is nature too complex to be truncated in a simple and short definition?

Cambridge Dictionary defines nature as: "*all the animals, plants, rocks, etc. in the world and all the features, forces, and processes that happen or exist independently of people, such as the weather, the sea, mountains, the production of young animals or plants, and growth*". The definition follows natural scientific knowledge of nature fairly well. Nature is more than just life, since abiotic circumstances play a crucial role in maintaining living organisms (Urry *et al*. 2017). This interaction is not just one-way as living creatures affect and alter their abiotic environment. Additionally, organisms affect each other via multiple interactions. The numerous complex direct and indirect interactions (in)between non-living and living matters construct ecosystems and their processes, which are necessary for existence of life (e.g. Hooper *et al*. 2005). In other words, the life itself together with non-living features enables its own existence. Hence, the refined version of the definition of nature in Cambridge Dictionary would read as follows: "*all the features, forces, and processes in the world which happen or exist because of all the living and non-living creatures and phenomena produced by them*".

1.1.2 Nature's contributions to species

None of the living organisms lives without Earth's resources. A resource, by its definition, is a species-specific matter. Depending on its life cycle, reproduction, diet, and other special characteristics, each species needs a certain amount of certain resources to thrive. Given that Earth's resources are finite, ecosystems are able to maintain certain levels of populations living on them. Ample resources allow population sizes to grow, but respectively, a lack of an essential resource causes competition between and within species that changes behavioral and physical responses of individuals and leads to slowing down or halting the population growth (Rockwood 2015). Population sizes are decreasing or increasing depending on the availability of essential resources. None of living organisms is endlessly able to exceed the carrying capacity of its environment.

Scientific literature provides numerous examples of species in various taxa showing population fluctuations due to the abundance or lack of resources. In the North Sea, a breeding success of a common guillemot (*Uria aalge*) is observed to significantly decrease when the population of a lesser sandeel (*Ammodytes tobianus*) is small, forcing guillemot parents to feed their chicks with other fish species including less energy (Wanless *et al*. 2005). In the Masai Mara's savannah ecosystem, the population of a wildebeest (*Connochaetes taurinus hecki*) dropped 81% in just 20 years from 1977 to 1997 mainly because their grazing, calving, and breeding areas were taken into the use of agriculture (Ottichilo *et al*. 2001). In the Easter Island, the former human (*Homo sapiens*) civilization fell apart in 1500th–1600th centuries likely as a result of the overuse of the island's slow-growing palm trees (Brander and Taylor 1998). Palm wood served as a firewood and material for making canoes and allowing fishing. Forests prevented soil erosion enabling effective agriculture and maintained bird populations that were hunted by residents. Loss of the forest cover likely led to resource depletion, finally causing collapse of the human society.

The limitedness of Earth's resources began to dawn on a modern historic human in the 16th century along with the Age of Discovery as Europeans sailed on the oceans and finally realized they had reached every continent of the world (Haila and Jokinen 2001). The first critique on thinkers of the Age of Enlightenment believing to an infinite progress of the human society was Thomas Malthus's essay where he stated that the human population would inevitably suffer from famine, since the population grows exponentially, but food production only linearly (Malthus 1798). The theory was incorrect, since technological development of food production has been rapid especially in the 20th century (Trewavas 2002), and since human population growth is estimated to cease in the 21st century (Raftery *et al.* 2012), but the presumption behind the theory, i.e. limitedness of resources was right. The reality of Earth's limited capacity to sustain human societies broke into the public awareness in 1972 when the Club of Rome published its report *The Limits to Growth*. The Club warned that the limits to growth would be reached during 100 years if growth of human population and resource depletion continued on its present trajectory, causing uncontrollable population decline and economic losses (Meadows *et al*. 1972), or in other words, immeasurable miseries.

During following decades, scientists began to propose that the benefits nature provides to people should be made commensurable with human-made goods and services in order to use Earth's resources wisely and sustainably (Westman 1977, de Groot 1987). Given that nature's services and functions are essential for economic growth and production, their value should be monetarized to allow politicians to compare benefits and costs of exploiting or preserving natural resources, and to make better-informed decisions. The concept of "Ecosystem Services" became established in scientific literature in 1990s (Costanza and Daly 1992, Perrings *et al.* 1992, Costanza *et al*. 1997) and in international politics along with the Millennium Ecosystem Assessment (2005). Ecosystem Services has grown into an independent academic research field, but it has been accused of a narrow ecological-economic view having a simplistic aim of setting a monetary value for all services provided by nature, including non-monetary and/or immaterial services (e.g. Chan *et al.* 2012, Lele *et al.* 2013). As a response to the critique of Ecosystem Services, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) launched a new concept of "Nature's Contributions to People" (Díaz *et al.* 2018). It is based on the concept of Ecosystem Services, but it acknowledges that culture is not just one dimension of the services as it was originally understood, but that it surrounds and permeates all the other services. Additionally, Nature's Contributions to People emphasizes earlier ignored or even disdained local and indigenous people's knowledge in determining nature's contributions.

As can be seen, a lot of work has been done in order to understand nature´s role in the well-being of human. The recent concept of Nature's Contributions to People unambiguously recognizes that "*nature is essential for human existence*" (IPBES 2019, p. 10). Despite understanding that without nature, human species will die, the work attempting to integrate nature's immeasurable value into the societies' practical functions such as decision-making or economic production seems challenging. Research evidence shows that we have significantly altered 75% of area of original terrestrial ecosystems, eradicated over 85% of wetlands' area, and directed so high pressures on the oceans that only 3% of their surface is free from human influence (IPBES 2019). When we hike in forests, sail on oceans, or climb to hills and mountains, most of us have an experience of being surrounded by intact nature. Truthfully, such nature barely exists anymore.

Vast impacts on ecosystems have naturally had substantial effects on species. Taxon-specific levels of an average risk of extinction varies greatly from less than 10% of bony fishes and gastropods to over 60% of cycads, but on average, one fourth of all species living on Earth are threatened (IPBES 2019).

They are under high extinction risk in the near future, or simply put, they can be lost forever during the next years or decades (Mace *et al.* 2008). Additionally, the biodiversity loss is predicted to continue and accelerate during next decades as a result of increase both in human population and economic consumption (Tilman *et al.* 2017, Powers and Jetz 2019).

Given that ecosystems' and species' functions and existence depend on each other (Hooper *et al*. 2005, Worm *et al.* 2006, Cardinale *et al.* 2012), it is not surprising that many ecosystem functions vital not only for human, but also for many other species are deteriorating. This is true especially concerning regulating functions such as climate regulation (Steffen *et al.* 2018, IPBES 2019) and pollinator diversity (Soroye *et al.* 2020).

As an exchange for altered ecosystems, declining regulating functions, and threatened species, we have gained huge amounts of food and materials. Since 1970, crops have increased three-fold and timber harvest 45% (IPBES 2019). Biomass of domestic poultry is nowadays three-fold compared to that of wild birds (Bar-On *et al.* 2018) and global gross domestic production four times larger than 50 years ago (IPBES 2019).

In short term, we are greatly improving Earth's capacity to support human societies, but in long term, we are greatly devastating them. In other words, we are not in the trajectory of sustainable development. Life on Earth demands resources. If human uses most of them, most of the other life must die.

1.2 Securing land for nature

1.2.1 Why to secure?

Following the definition of nature adapted from Cambridge Dictionary, human societies are a part of nature. Then, why protected areas, i.e. areas where land-use and many other human activities are forbidden or controlled (Dudley 2008), should be established? What we protect against and for whom?

The answer depends on the chosen environmental ethics. Biodiversity supports the supply of many ecosystem services (Mace *et al.* 2012, Gamfeldt *et al.* 2013) and promotes achieving the goals of sustainable development such as poverty alleviation (Blicharska *et al.* 2019). Establishing and maintaining protected areas is a direct and an efficient way to slow down and halt habitat loss (Geldmann *et al.* 2013) and decline of species' populations (Gray *et al.* 2016). According to the anthropocentric ethics that places an instrumental value on nature (Vucetich *et al*. 2015), the ability of protected areas to maintain Earth's biodiversity is important because they simultaneously help to maintain the human societies (Rands *et al.* 2010). From this point of view, nature conservation can be seen as a mean to protect humans from their own harmful actions.

Another ethical viewpoint comes from the ecocentric ethics arguing that nature has an intrinsic value (Soulé 1985). Accordingly, nature should be valued for what it is instead of what it does, and treated with respect for this (Vucetich *et al*. 2015). Hence, all forms of life deserve to exist independently from their instrumental value. Since nature conservation supports these aspirations, it can be seen to protect biodiversity from human exploitation.

1.2.2 Why securing is hard?

International community has acknowledged both the instrumental and intrinsic value of nature (e.g. CBD 1992, IPBES 2019). The Convention of Biological Diversity, signed by the majority of the world's countries, sets the Aichi Biodiversity Target 11 that aims to protect 17% of terrestrial and inland water areas, and 10% of coastal and marine areas by 2020 (CBD 2011). The global quantitative target for both terrestrial and marine environments will likely be fulfilled, but regional differences in coverage of protected areas are huge and management and resource allocation often insufficient (Gannon *et al.* 2019), which seriously hamper protected areas' ability to fill their duty in maintaining biodiversity (Geldmann *et al.* 2018, Jones *et al.* 2018).

There are voices arguing that sustainable use of natural resources, i.e. the use simultaneously safeguarding nature and maintaining socio-economic development (United Nations General Assembly 1987), is impossible since in most cases, the needs of nature and human unavoidably contradict (Robinson 1993). More widely speaking, each individual's needs contradict with its environment as we all need to use resources to keep alive and in the finite world, one's resource use often happens to the detriment of other's. In the light of empirical evidence, biodiversity protection and satisfying human needs more often contradict than do not (McShane *et al.* 2011). Examples of biodiversity protection surrounded by conflicts and compromises are numerous. For instance, throughout Europe, the conservation program Natura 2000 established by the European Union has caused conflicts and raised direct rage between conservation authorities and local land users (e.g. Alphandéry and Fortier 2001, Hiedanpää 2002, Grodzinska-Jurczak and Cent 2011, Blondet *et al.* 2017). In tropical developing countries, establishing areas of strict protection have been implemented by displacing hundreds of thousands of people from their homes or alternatively, by setting considerable restriction for resource use, both causing poverty and other social problems (Adams and Hutton 2007, Lele *et al.* 2010). Even attempts to design programs that aim to improve both the state of biodiversity and well-being of local people have usually failed at least partly: either they have not been able to protect ecosystems in a projected manner, or they have not met the needs and expectations of local residents (Lele *et al.* 2010, McShane *et al.* 2011, but for a potential exception, see Nelson *et al.* 2009).

Indeed, biodiversity protection is hard although its benefits are multiple and often invaluable. Perhaps the most notable reason for this is that benefits and costs of protection for people do not spread evenly or equally in spatial, nor temporal terms (e.g. Balmford *et al.* 2002, Balmford and Whitten 2003, Soares-Filho *et al.* 2010, McShane *et al.* 2011). Instead, mismatches exist between a local, a national, and a global level, and in short and long term.

From the economic point of view, direct costs of establishing protected areas such as practical conservation planning or acquiring land for protection are often borne by governments or national or global communities, whereas indirect costs such as opportunity costs of using protected land for other purposes often fall on local people (Balmford and Whitten 2003, Kabii and Horwitz 2006). Economic benefits, by contrast, are often disseminated relatively more on a global or a national level than on a local level (Balmford and Whitten 2003). Additionally, economic profit of land development often materializes fast, making development rational in short term, but potentially undermining the long-term sustainability of land use, therefore impairing possibilities for future economic operations (Balmford *et al.* 2002). Instead, benefits of setting land aside for protection often materialize slowly and/or are harder to demonstrate than those of land development.

Economic considerations are not the only reason for conservation conflicts. Social matters often play an important role (e.g. Kamal *et al.* 2015, Olive 2016). Although protection would not substantially affect income or living standards of land users or owners, they can lose their traditional livelihoods and/or cultural practices because access to protected land or to its resources is blocked or significantly impeded (Vedeld *et al.* 2012, Bennett and Dearden 2014, De Pourcq *et al.* 2017). They can feel socially excluded from their land due to insufficient participation in decision-making processes (Lele *et al.* 2010, Brondo and Bown 2011, Blicharska *et al.* 2016, De Pourcq *et al.* 2017). Even though access to the land would stay unchanged, landowners can feel their property-rights being insulted and protection being unfair and unequitable (Jackson-Smith *et al.* 2005, Kabii and Horwitz 2006, Kamal *et al.* 2015b, Olive 2016).

The dilemma between biodiversity conservation and extraction of natural resources for human well-being can be seen as a question of the tragedy of the commons (Hardin 1968). Spatial and temporal mismatches of benefits and costs acquired from protection vs. land development make sure that often the latter is seen more beneficial than the former (e.g. Balmford *et al.* 2002, Balmford and Whitten 2003). Similarly in Hardin's (1968) seminal paper, single herdsmen benefitted from adding heads to the livestock, whereas costs in the form of overgrazed pastures were distributed over all herdsmen.

Eventually, in order to make nature conservation an appealing action, it should be profitable and cost-effective not just for the society, but also for the individuals. The task is not easy in the world of increasing human population, growing per capita consumption (Steffen *et al.* 2015), and exacerbating land scarcity (Lambin and Meyfroidt 2011).

1.2.3 Who should secure?

Given the peculiar characteristics of nature that make benefits and costs of its protection to distribute unevenly in space and time and between people, one can see governments' lands as self-evident locations for protected areas. Governments do provide also other services that are hard or impossible to provide by private agents such as water supply, sewerage, or national defense.

Currently, the protected area network is both globally and nationally biased on areas that are distant, of low productivity, and not under high development pressure (Joppa and Pfaff 2009), whereas global biodiversity hotspots are largely unprotected (Jenkins *et al.* 2013). Biodiversity would be most efficiently safeguarded by intergovernmental conservation planning cooperation as it would reduce the total protected area and lead to a more connected conservation network compared to national planning (Pouzols *et al.* 2014), but as long as global planning is not made, conservation actions need to be implemented in every regions. Otherwise, the responsibility of protection is just transferred to somebody else and without efficient international coordination, there is a danger that no one will protect.

One of the tricky global conservation challenges is that plenty of the most valuable and endangered biodiversity lies on private hands (e.g. Knight 1999, Norton 2000, Stephens 2001). Therefore, if biodiversity is desired to be maintained and its benefits to be enjoyed, private land protection is probably inevitable. In the next two chapters, I will summarize approaches for private land protection divided according to their obligatoriness.

1.2.4 Involuntary conservation approaches

Involuntary conservation means that the government expropriates land for conservation purposes without providing a possibility for landowners to affect a conservation decision (Kamal *et al.* 2015a). Commonly, this procedure is applied for many societally important purposes such as large infrastructure initiatives. Given that the current global environmental challenges can threaten the human societies in their current form (IPBES 2019), using expropriations for conservation purposes could be seen reasonable. Originally, however, the aim of land expropriations in nature conservation was not to safeguard existence of biodiversity or a humankind. Instead, the aim was, in a romantic and elitist spirit, to create places where people could refresh their soul contaminated by civilized life and feel emotions evoked by the wilderness (Cronon 1995, Colchester 2004). Hence, the original premise of expropriations for conservation purposes has been quite different from that of most other societally important projects.

In the most restrictive form of involuntary conservation, local inhabitants are displaced from their homes. Roots of this method are brutal: they date back to the 19th century, when the first protected areas and national parks such as Yosemite and Yellowstone were established in USA, following slaughters and evictions of indigenous people (Colchester 2004). In the 20th century, the method of displacing locals and preventing their traditional livelihoods was exported to other continents and applied regularly especially in developing countries, causing many social problems (Colchester 2004, Adams and Hutton 2007, Lele *et al.* 2010).

Another approach of involuntary conservation is to expropriate land without displacements, but to set certain land management restrictions in order to prevent actions that could deteriorate biodiversity values (Kamal *et al.* 2015a). Even if these actions would not significantly harm landowners in economic or cultural terms, they can provoke feelings of unfairness and property rights violence (Jackson-Smith *et al.* 2005, Kabii and Horwitz 2006, Kamal *et al.* 2015b, Olive 2016).

At least one negative side effect is probably quite distinctive to involuntary conservation approaches. Pre-emptive behavior refers to actions that landowners implement intentionally, aiming to destroy biodiversity values existing on their lands in order to prevent land use regulations caused by conservation (Brook *et al.* 2003, Lueck and Michael 2003, Zhang 2004, Jokinen *et al.* 2018, Simmons *et al.* 2018a). These actions can be illegal killing of certain species or logging, burning, ditching, or otherwise harming habitats colonized by them. Pre-emptive actions are not always directed on certain species since they can also be landowners' responses to generally tightening environmental regulation. Pre-emptive behavior as a phenomenon is widely known, but little studied.

Compensation practices of involuntary approaches vary a lot depending on the country and on the case. For instance, the Endangered Species Act (ESA) in USA have prohibited taking of species listed in the Act and harming their critical habitats since 1973 (Endangered Species Act of 1973), but practices supporting landowners, whose lands host listed species, were started to develop not until 1985 (Byl 2019). They include releases to land development restrictions and assurances of not to set more restrictions by environmental authorities, if the species is generally doing well on the land property in question (Donahue 2005). In Finland, by contrast, landowners are economically compensated regardless of whether their lands are protected against landowners' will or on a voluntary basis (Nature Conservation Act 1096/1996).

1.2.5 Voluntary conservation approaches

In voluntary nature conservation, a landowner decides whether to engage to or opt out from conservation actions (Kamal *et al.* 2015a). Governments, non-governmental organizations (NGOs), and other agencies facilitate voluntary conservation actions by providing landowners with various incentives, which content, support, and validity varies a lot. They can offer e.g. direct payments, tax reliefs, land exchanges, or guidance as an exchange e.g. for a land title, for relinquished land development rights, or for a landowner's commitment to implement certain actions that maintain or improve biodiversity values.

In some countries, the history of voluntary conservation is almost as long as that of the involuntary one. For instance, in USA and New Zealand, the first land trusts purchasing land for conservation from voluntary landowners were established in the end of the 19th century (Wright 1992, Lochhead 1994). However, while some large and influential land trusts such as the Nature Conservancy in USA had lived their golden age already for several decades, voluntary conservation approaches took barely their first steps in some other countries. In Finland, for instance, the first voluntary forest conservation program called METSO, was started in the millennium, quickly getting support from various stakeholders including conservation organizations, forestry agencies, and landowners' interest groups (Hiedanpää 2005, Paloniemi and Vilja

2009). Indeed, voluntary conservation has globally succeeded in alleviating land use conflicts. Its success lies in ensuring private property rights and fairness, and engaging landowners in a conservation process, which enhances trust and knowledge-building between different parties and emphasizes landowners' ability and will to be good stewards of their land (e.g. Kabii and Horwitz 2006, Paloniemi and Tikka 2008, Kamal *et al.* 2015b, Olive 2016, Young *et al.* 2016).

However, voluntary conservation approaches can be inefficient in terms of ecology and economics. They do not always fulfill conservation targets (Guerrero *et al.* 2010, Von Hase *et al.* 2010, Knight *et al.* 2011, Adams *et al.* 2014). In order to better achieve the targets, the number of conservation-willing landowners needs to be increased, which requires more financial resources. Even with larger resources, voluntary conservation approaches may not achieve the same level of protected biodiversity as could be achieved, if all land was available for protection regardless of landowners' opinions (Lewis *et al.* 2011).

Generally, it seems that involuntary conservation approaches can cause more social harm than voluntary ones, whereas voluntary conservation approaches can be ecologically and/or economically inefficient compared to involuntary ones. It is very hard to evaluate, which one produces larger net gains for nature in long term. Only one thing seems sure in the struggles to safeguard biodiversity: whether conservation is implemented by involuntary or voluntary means, to protect is better than not to protect, since land use pressures and the risk of extinctions are projected to increase in the future (Lambin and Meyfroidt 2011, Tilman *et al.* 2017).

1.3 Conservation decision-making

1.3.1 Biodiversity as a spatial phenomenon

Communities, populations, and individuals of living organisms are always located in particular ecosystems and habitats where abiotic factors such as climate, light, or nutrients, and biotic factors such as food resources, competition, or predation enable them to occur (Urry *et al.* 2017). Therefore, protection of biodiversity needs to take place where biodiversity locates. Nevertheless, since biodiversity protection is just one form of land use, it unavoidably competes with others (Lambin and Meyfroidt 2011). Indeed, Earth is crowded in terms of land use practices, each of which are affected by numerous social and economic matters. Therefore, multiple other factors than the location of biodiversity have an effect on where the protection is finally implemented (Knight and Cowling 2007). These include e.g. a potential for natural resource extraction (Moilanen *et al.* 2011), different threats imperiling biodiversity (Wilson *et al.* 2011, Tulloch *et al.* 2015), land price or opportunity costs (Adams *et al.* 2010, Wilson *et al.* 2011, Mazor *et al.* 2014), preferences of citizens owning or utilizing the land (Guerrero *et al.* 2010, Knight *et al.* 2011), and complexities in decision-making processes (Kareksela *et al.* 2018). In summary, due to its spatial nature and its positioning in complex social-ecological systems, biodiversity conservation is often surrounded by wicked problems concerning its planning, decision-making, and implementation.

1.3.2 The science-practice gap

Science is expected to produce knowledge and resolutions to societal decision making, but the inability of integrating science to practice is a well-known problem in many disciplines, including also conservation planning (e.g. Sutherland *et al.* 2004). A wide array of literature illustrates reasons for the science-practice gap (also known e.g. as a research-implementation, knowing-doing, or theory-practice gap). For instance, the scientific knowledge is too abstract, uncertain, or controversial, or inadequately translated to practitioners and decision-makers, and the scientific research is made on wrong scales or studies irrelevant questions in relation to practical conservation (e.g. Knight *et al.* 2008, Hulme 2014, Bertuol-Garcia *et al*. 2018).

Spatial prioritization methods have been developed as one tool to narrow the science-practice gap as they enable resolving practical conservation problems in a scientifically rigorous and transparent manner (Moilanen *et al*. 2009c). They provide explicit solutions and material to political conservation decision-making and can be used to review ecological, economic, and social impacts of made decisions. For decades, attempts to support practical conservation planning via spatial prioritization seemed not to pay off as most of the prioritization analyses presented in peer-reviewed literature did not contribute to conservation actions and many of the scientists did not even aim that (Knight *et al.* 2008). The recent research shows, however, that the situation is changing: over two-third of prioritizations meant to be implemented in practice also did so, and the collaboration between scientists and practitioners seems to become a more general practice (Sinclair *et al.* 2018). Instead, compared to past years, the scientific conservation planning research hardly better contributes to practical conservation implementation: most of the research is still focused just on ecological factors, utilizing socio-political research is rare despite its acknowledged importance in making successful conservation, and research concerning monitoring and implementation has even declined during the 21st century (Mair *et al.* 2018). It seems that spatial prioritization as a technical tool is finally about to deliver its goods in directing practical conservation actions and filling the science-practice gap, but research concerning conservation planning as a wider phenomenon still does not fulfil its potential.

1.3.3 Structured Decision Making framework

Value judgements are an integral part of nature conservation, whether it is a question of setting ecological targets for conservation, allocating resources, or deciding how much trouble conservation is allowed to cause to private landowners (e.g. Wilhere 2008, McShane *et al*. 2011). Structured Decision Making is a framework which aims to inform and assist decision-makers in decisions that are shaped with value- and trade-off-related contradictions (Gregory *et al*. 2012). It suits well for managing public resources, since it focuses on assisting groups including diverse range of stakeholders by emphasizing the use of transparent and defensible decision structuring approaches. The premise in Structured Decision Making is that decisions should be based on understanding values and consequences. In other words, decision-makers need to define what is important, and evaluate the consequences of alternative decisions. Consideration and evaluation of alternatives enable finding the best solution for the problem at hand, but they also make value judgements visible both for persons making judgments and for persons evaluating the alternative choices. Conservation often requires making decisions that cause harm for some actors or feel somehow "wrong" (McShane *et al*. 2011, Batavia *et al*. 2020). Evaluation of alternative consequences makes these hard choices easier (Gregory *et al*. 2012). The aim in Structured Decision Making is not to produce any particular result and it cannot guarantee a successful outcome, but it provides a reasonable framework where decision-makers are involved in good decision-making practices (Gregory *et al*. 2012).

1.4 The ecological and societal context of mire conservation in Finland

1.4.1 Peatland use

Small-scale utilization of peatlands for agricultural use in Finland dates back to prehistoric time (Vasander 2006). Sedge-dominated lakesides and riverbanks were used for haymaking and as pastures, and especially in the northern Finland, spring floods were directed on mire meadows by damming in order to prevent growth of tree seedlings and to promote growth of herbs. In the 18th century, a peatland cultivation technique was for a first time described in a written form. Taking peatlands for agricultural use became more common especially in the southern and western parts of Finland, and first drainage operations were made in the 1860s during the Great Famine Years to increase the area of arable land. In the 19th century, peatlands were started to be perceived as wastelands that should be modified for utilization wherever possible. The Finnish Peatland Cultivation Association, established in 1894, actively inventoried peatlands suitable for agriculture, peat mining, and forestry up to 1950s.

Wooded mires, similarly as mineral soil forests, have presumably been a source of firewood and other small-scale household use for thousands of years. Wide and systematic use of peatlands for forestry purposes started not until in the beginning of the 20th century (Lindholm and Heikkilä 2006). The first remarkable episode towards intensive peatland forestry was enacting the Forest Improvement Act in 1928, which aim was to organize financial resources to drain peatlands for forestry purposes (Law for reserving resources… 140/1928). Few production launched massive and systematic peatland drainage funded by the Finnish government and the International Bank for Reconstruction and Development (Palosuo 1979). During several years, hundreds of thousands of peatland hectares were annually ditched (Lindholm and Heikkilä 2006). Reasons for the very intensive ditching operations were at least partly understandable: the peace treaty made with the Soviet Union after the Second World War required Finland to pay large reparations where an effective forestry played a significant role. Generally, the societal atmosphere desired to improve the well-being of the impoverished nation. Indeed, the actions to increase wood production were successful: the growing stock volume of Finnish forests was 1.8-times larger in 2018 than in 1921 (Statistical database of Natural Resources Institute Finland: https://urly.fi/1sLm). Active ditching and improved forestry practices such as fertilization and breeding of saplings are the most important reasons for the development (Haapanen *et al*. 2015, Moilanen *et al.* 2015, Henttonen *et al.* 2017).

Large-scale ditching of pristine mires was ceased before the millennium (Lindholm and Heikkilä 2006). However, mires are still cleared and/or ditched e.g. for forestry (Kaakinen *et al*. 2018), peat mining (Ministry of Agriculture and Forestry 2011), and agricultural purposes such as spreading livestock manure (Regina *et al.* 2016). Though new ditches are not officially excavated anymore, one can find recently logged wooded mires that are newly ditched in order to ease tree stand regeneration (in Finnish "ojitus- tai naveromätästys") (Fig. 1).

FIGURE 1 A recently logged and ditched spruce mire in eastern Finland in 2019. One can see from maps that the mire was in a pristine state before the harvesting and ditching operations. Officially, new ditches are not excavated anymore on pristine mires, but in practice, they can be found on recently harvested wooded mires. Photo: Eini Nieminen.

Original area of peatlands in Finland was about 10.4 million ha (Kaakinen *et al*. 2018). Due to utilization of peatlands, about 1.7 million ha (16%) have been lost and are not classified as peatlands anymore. Of the remaining 8.7 million ha, 4.6 million ha (60%) have been ditched, mostly for forestry purposes. Percentual proportion of ditching varies regionally: in northern Lapland it is less than 15%, but in the majority of southern Finland over 75% (Alanen and Aapala 2015). All the 4.1 million ha (40%) of non-ditched peatland area is not in a pristine state as forestry operations are made on about third of them, strongly focusing on southern Finland (Kaakinen *et al*. 2018).

1.4.2 Peatland protection

In the 1970s, protected mire area was 193 000 ha of which 150 000 ha was protected by the decision of Metsähallitus (the organization governing the government-owned lands in Finland) and 43 000 ha by legislation (Ruuhijärvi 1978). The need for complementary protection was acknowledged and the preparation of the Basic Plan for Peatland Preservation started. In 1979 and 1981, the most urgent parts of the Basic Plan were implemented, increasing the protected area by about 100 000 ha. The Basic Plan focused on protecting large and watery mires having well-known avifauna, but uncharacteristic, rare, and small mires, as well as mires not belonging to any geomorphological mire complex were ignored (Alanen and Aapala 2015).

The national evaluations of threatened species and habitats show that mires' biodiversity is declining. 54% and 76% of evaluated mire habitats are either threatened or their future trend is declining, respectively (Kaakinen *et al*. 2018). 10.6% of mire species are threatened, almost half of them living on rich fens (Hyvärinen *et al*. 2019). The state of mire associated biodiversity is worse in southern than in northern Finland.

As a response to declining mire biodiversity, Council of State made a decision about expanding the mire conservation network in 2012 (Council of State 2012). The aim was to enhance the protection of a representative mire network across different vegetation zones in Finland (Fig. 3 in Methods) by protecting approximately 100 000 ha of mires. The Complementary Mire Protection Program (hereafter CMPP) was politically decided to be prepared and implemented as a conservation program based on the Nature Conservation Act (Council of State 2012). In practice, the Act enables the government to expropriate land for conservation purposes in such cases where a landowner is not willing to protect. The Act also obligates the government to pay an economic compensation to landowners of the protected sites, whether the sites are protected voluntarily or via expropriations. Owners of the lands chosen for protection are allowed to decide whether to keep the ownership of the land resulting in a private conservation area, or to sell it to the government. In both cases, landowners are compensated by being paid a market price for their land, or by exchanging their land for an equivalent parcel of the government's land, depending on the landowner's will.

In 2014, the CMPP preparation work was almost ready and practical implementation approaching (Salomaa *et al.* 2018). At that time, the Green Party hold a position of the Minister of the Environment. However, the Green Party left the government because the coalition partners granted a license for a nuclear power plant. New Minister of the Environment was appointed from the Coalition Party. During the first weeks of her term, the new Minister ceased CMPP preparation and launched an account about options to protect mires from a voluntary basis. In the end of 2014, the Minister decided of the voluntary implementation of CMPP and rejection of the option for land expropriations. The working group responsible for the CMPP preparation made a suggestion to protect about 117 000 ha of mires (Alanen and Aapala 2015). In 2015, the Center Party won the parliamentary election and the new government decided to cut heavily the nature conservation budget. Due to the budget cut and the lack of administrative tools to protect mires from a voluntary basis, CMPP implementation was stagnated apart from some government-owned mires and some such privately-owned wooded mires that were able to be protected in the voluntary forest conservation program METSO.

Environmental authorities, conservationists, and conservation researches acknowledged that the original plan to enable land expropriations in CMPP implementation could provoke pre-emptive loggings on candidate wooded mires in order to avoid their protection. Messages referring to such actions can be found in social media such as Twitter and internet forums of forestry magazines (Fig. 2). However, rejecting the option of expropriations did not ease off the concern of candidate wooded mires ending up to be harvested. Forestry in Finland is so intensive that majority of forest sites are logged once they reach maturity (Natural Resources Institute Finland 2019). Moreover, harvesting of sites considered to be protected is legal. Since voluntary mire conservation was known to lengthen the time lag between the consideration for protection and the actual conservation actions, the rejection was thought to increase the likelihood of candidate wooded mires to be harvested simply because there would be more time for loggings to happen.

In 2019, the government led by the Social Democratic Party started to return conservation appropriations (Council of State 2019). In the beginning of 2020, Ministry of the Environment launched the habitat conservation program Helmi, which aims to support biodiversity in Finland and improve the state of habitats e.g. by protecting and restoring mires, habitats of waterfowls, traditional rural biotopes, forest habitats, and shores (Ministry of the Environment 2020). Concerning mires, the aim is to protect 20 000 ha by 2023 on a voluntary basis and by compensating the economic losses for landowners (https://www.ym.fi/helmi). Currently, the political atmosphere is such that the option of land expropriations based on the Nature Conservation Act will not be applied in the near future.

In the beginning of 2020, before starting the Helmi-program, the mire conservation network had been expanded by about 40 000 ha, focusing strongly on government-owned mires.

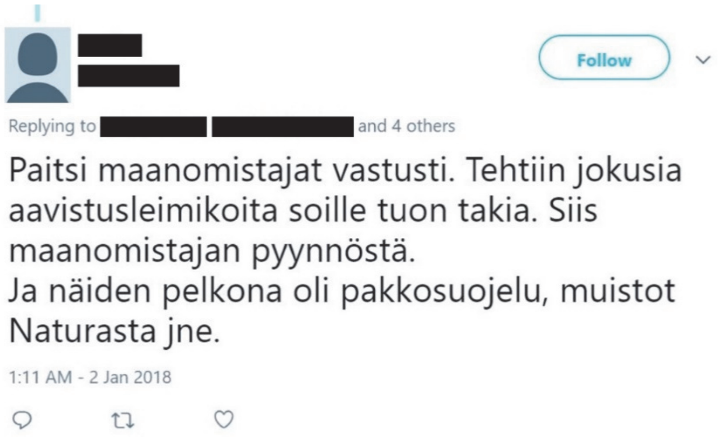


FIGURE 2 An example of a tweet claiming that preparation of CMPP would have caused pre-emptive loggings on candidate wooded mires. Translation in English: "Except for landowners who resisted [CMPP]. Some stands were marked for pre-emptive cutting because of it [CMPP]. For a landowner's request. And they feared for forced protection, and remembered Natura [Natura 2000, the European-wide conservation program planned and implemented as a top-down procedure by European Union]."

1.4.3 Peatlands as conservation subjects

Peatlands' biodiversity and their function as ecosystems depends on a water table level and on wooded mires also on tree stand (Moore and Knowles 1989, Laine *et al.* 1995, Jungkunst and Fiedler 2007, Haapalehto *et al.* 2011, Maanavilja *et al.* 2014). Water table level can be affected e.g. due to ditching or forest loggings occurred on the peatland's catchment, causing hydrological changes and disturbing or changing the function of an original peatland ecosystem (Tahvanainen 2011). Due to peatlands' dependence on water, their conservation can be effective in long term only when their hydrological processes are safeguarded which often means setting aside relatively large continuous areas. On many peatlands, land use pressures are or have historically been high so often they are degraded and/or exposed to an intensive competition between different forms of land use (Vasander *et al.* 2003, Pin *et al.* 2011). Furthermore, in many countries such as UK and Finland most of the peatlands are privately owned (Bain et al. 2011, Alanen and Aapala 2015). For these reasons, protecting large continuous areas can be challenging in terms of economy and/or social acceptability.

1.5 Aims of this thesis

The society causes many challenges on nature conservation. My aim is to study some of them and find solutions that could ease contradictions between the needs of the society and non-human nature. I implement the research in Finland in the context of boreal mire conservation.

In the first chapter (I), the aim is to understand which roles and possibilities spatial prioritization can provide for conservation planning and decision-making. I describe a prioritization analysis and a decision-making context that support and benefit each other. Here, the political premise for mire protection was to allow land expropriations for conservation purposes based on the Nature Conservation Act. This invoked rumors, claims, and fears that landowners would pre-emptively log their wooded mires in order to prevent protection. Therefore, in the second chapter (II), I study whether the wooded mires considered for protection in CMPP were pre-emptively logged.

Due to unforeseeable political changes, the option for expropriations was withdrawn just before the practical implementation of CMPP was about to start. Withdrawal allowed conservation-resistant landowners to refuse from protection of their lands. This restricted mires available for protection and changed the prioritization problem remarkably, causing a mismatch between the prepared CMPP conservation plan and the new political requirement. Hence, in the third chapter (III), I proceed to study which kind of effects voluntary conservation would have on mire protection (conservation scenario 1) compared to the situation where land expropriations would be allowed, i.e. all the candidate mires originally considered for protection would be available (conservation scenario 2). Then I develop a prioritization model that builds such mire conservation network that strives to avoid protecting lands of conservation-resistant owners, but still to pick mires of high conservation value and low financial costs (conservation scenario 3). Finally, I compare these alternative conservation scenarios in terms of average representation of biodiversity features included in the protected area network, the financial costs of conservation, and the amount of landowners' resistance to protection of their land.

The research questions in chapters I–III are:

1. How to build a spatial prioritization analysis to take into account multiple ecological and societal needs in the context of mire protection? (I, III)
2. How to make spatial prioritization and conservation decision-making to support each other and fill the science-practice gap? (I)
3. Do the harvesting rates of wooded mires chosen as the candidate sites for boreal mire protection differ from the harvesting rates of similar wooded mires that were not considered for protection? (II)

4. Which effects acknowledging vs. ignoring landowners' conservation resistance can have on ecological gains and financial conservation costs in the context of boreal mire conservation? (III)

2 METHODS

2.1 The coverage of CMPP

CMPP covered the whole of Finland except for northern Lapland and the Åland Islands in the south-west (Fig. 3) (I, III). In the CMPP preparation, 1533 unprotected candidate mires (327 300 ha) and 3400 already protected mires (601 700 ha) were considered.

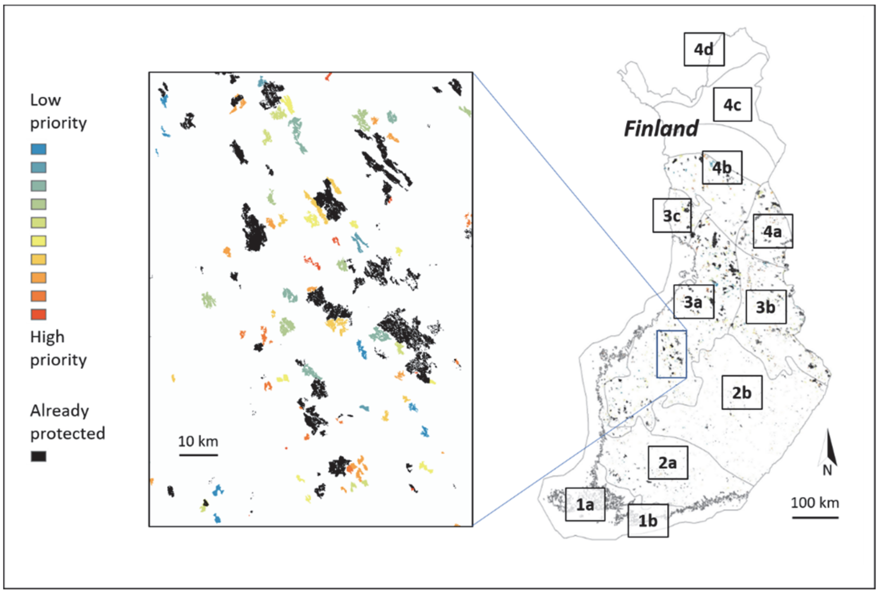


FIGURE 3 Mires considered in the CMPP preparation. The existing protected mires are represented in black color and the candidate mires in a color scale from red to blue according to their priority order produced in chapter I. Vegetation zones are shown (1a Hemiboreal, Åland; 1b Hemiboreal, oak zone; 2a Southern boreal, Southwestern Finland and Southern Ostrobothnia; 2b Southern Boreal, Lake District; 3a Middle Boreal, Ostrobothnia; 3b Middle Boreal, Northern Carelia-Kainuu; 3c Middle Boreal, Southwestern Lapland; 4a Northern Boreal, Kuusamo District; 4b Northern Boreal, North Ostrobothnia; 4c Northern Boreal, Forest Lapland; 4d Northern Boreal, Fjeld Lapland). Northern Lapland and Åland Island were excluded from the CMPP preparation. Adapted from chapter I, i.e. Kareksela *et al*. 2020.

2.2 Prioritization method

Spatial prioritization encompasses two main methodological families: integer linear programming and heuristic optimization (Moilanen and Ball 2009). Integer linear programming methods are able to find the most optimal conservation solution with feasible computing times when the problem in question is not very large and can be reliably linearized (Haight and Snyder 2009, Beyer *et al.* 2016). Instead, heuristic methods are able to solve very large prioritization problems that have non-linear and stochastic characteristics, but unlike in integer programming methods, there is no guarantee of the solution's optimality (Moilanen and Ball 2009). Therefore, both the methods are suitable, but for different problems (Moilanen 2008).

Zonation is a heuristic systematic spatial prioritization method and software that helps to identify sets of sites where connectivity and quality best retains biodiversity features in long term, given different limitations such as costs or threats (Moilanen *et al*. 2005, Moilanen *et al*. 2014). It utilizes grid-based spatial data of e.g. observed or predicted occurrences of species and habitats, acquisition or opportunity costs of certain areas, potential of the areas for different purposes such as natural resource extraction or carbon sequestration, or basically anything that can be translated into spatial form. In practice, Zonation perceives the input data (i.e. single grid cells or groups of grid cells called planning units) as a landscape that hosts the biodiversity features included in the data, and assumes that the whole landscape in question is protected (Lehtomäki and Moilanen 2013). It then starts to iteratively remove sites which deletion causes the smallest aggregate marginal loss for biodiversity in the whole landscape. The removal proceeds until all sites are removed. As a result, Zonation produces a hierarchical nested ranking of all sites in the landscape so that the best 2% of the sites is included to the best 5%, which are included to the best 10%, and so forth. Hence, Zonation differs from other well-known prioritization software such as Marxan (Ball *et al*. 2009) in that it does not require pre-defining conservation targets but allows observing how representation of biodiversity features or other factors included in the analysis changes while protected area increases (Moilanen *et al*. 2009a). Therefore, Zonation suits well for studying interrelationships and relative changes of ecological, economic, and social factors included in the prioritization analysis.

2.3 Preparing the complementary mire protection

2.3.1 Phases of the preparation

The working group responsible for CMPP preparation comprised 14 stakeholders and experts, including mire ecologists, land-use planners, and conservation scientists, and representatives from the environmental and forestry administrations, the landowners' association, and conservation NGOs (I). The working group concluded that including a spatial prioritization analysis to the planning process of CMPP would enable complementarity-based selection of mires and taking into account mires' ecological connectivity. The prioritization analysis was from the beginning designed in close cooperation between the working group and prioritization experts (I). In addition to the complementarity-based prioritization, the working group also created a scoring system for the candidate mires that assisted the expert work in selecting the mires. Prioritization analysis and the scoring method together formed a Structured Decision Making process (Gregory *et al.* 2012) (I).

2.3.2 Data

Spatial data was based on a field survey of the candidate mires, a habitat database of protected areas, small water bodies picked from topographic database, modelled likelihood of territories of mire associated bird species, and species observation data from the Finnish threatened species database (I). Altogether 91 spatial data layers were collated on geomorphological mire complexes (31 layers), mire habitats (39), threatened plants and mosses (3), small water bodies (1), and potential habitats for mire associated birds (17). As condition data for the mires, spatial data of ditches on peatlands from topographic database was applied. All spatial data were converted to raster data layers having a resolution of 50 x 50 meters.

2.3.3 Prioritization analysis

Prioritization analysis was performed using the Zonation software (Moilanen *et al.* 2005, Moilanen 2007, Moilanen *et al.* 2014) (I). Analysis was constructed in stages in order to observe how adding different data and analysis approaches affected the priority of mires. The analysis structure was as follows (I):

- Biodiversity features were weighted based on their Red List status across the whole planning area and more detailed regional Red List statuses (Raunio *et al.* 2008, Rassi *et al.* 2010).
- Hierarchical mask layer was used to separate the present mire conservation network and the candidate mires in order to identify which candidate mires best complement biodiversity of the existing mire conservation network (e.g. Mikkonen and Moilanen 2013, Virtanen *et al.* 2018).
- Planning units (groups of grid cells) were used to enable prioritization of the mire ecosystem complexes as hydrological entities.
- Condition layer was applied to de-emphasize areas where land-use pressure has led to a loss of ecological condition. The condition layer prioritizes mires that are comparatively less damaged due to drainage (Kareksela *et al.* 2013, 2018).

- Administrative unit analysis was used to balance local and national scale rarity and weighing of the biodiversity features (Moilanen and Arponen 2011). This simultaneously considers both regional and national priorities, leading to more spatially balanced distribution of the top priority mires among the considered regions.
- To emphasize the ecological connectivity of mires, the method of interaction connectivity was applied (Lehtomäki *et al.* 2009, Rayfield *et al.* 2009). This increases the priority of candidate mires that have other ecologically high-quality mires nearby. Connectivity was emphasized between all mires included in the analysis, but more for the mires already having a protection status.

The prioritization results were processed with ArcMap 10 (I).

2.3.4 Integration of expert knowledge

The working group needed to agree on prioritization settings to address the problem of selecting mires for protection, e.g. by defining what elements such as data, connectivity, or regional priorities they wanted to include in the prioritization, and how each element was to be emphasized. Therefore, the prioritization analysis provided also as a platform for the expert knowledge use. The co-production of the analysis was carried out in stages (I):

1. The working group outlined the detailed targets that would best serve the goal of CMPP. (It should be noted that Zonation as a prioritization method does not require habitat or species-specific targets for biodiversity feature representation, but it instead aims to cumulative persistence of the most complementary biodiversity features).
2. Alongside with building the prioritization analysis, its performance was monitored (Figs. 5 and 6) by adding the key parameters one by one.
3. Zonation produces performance curves that describe representation levels of biodiversity features included in the analysis. Using the curves, the prioritization experts visualized to the working group how the biodiversity coverage continuously improves when new mires are added into the mire conservation network. Typically, gains are highest with the first additions and level off, i.e. become saturated when making lower priority additions (Fig. 5). This is because the first site additions serve as biodiversity gap fillers. i.e. they rapidly add coverage for many narrow-range features that have missing or low representation in the existing protected area network (Sharafi *et al.* 2012, Virtanen *et al.* 2018). The level of saturation was used to decide how large area should be chosen according to the prioritization analysis and how much to choose based on the candidate mire scoring and expert knowledge.
4. The working group qualitatively examined candidate mires that were proposed for protection according to the prioritization analysis. The aim was to make sure that no “strange choices” existed. For instance, some of

the very small or recently drained mires were replaced with more representative ones.

5. The actual mire selection was then carried out hierarchically. First, according to the prioritization analysis, mires best complementing the existing mire protection network were chosen. Then, based on the scoring method, the mire conservation network was further complemented with the highest scoring mires within each administrative unit.

2.4 (Non)existence of pre-emptive loggings on wooded mires

2.4.1 Timing of the CMPP events

Council of State made its decision of CMPP preparation in August 2012 (II). The public informing of CMPP started in the beginning of 2013 by announcements in newspapers, a poll in a government operated citizen portal in the internet, and hearings of stakeholder representatives. In May–July 2013, landowners of the candidate mires received personal information letters notifying about the field inventories that were made for the preparation of CMPP during the summer 2013 (Alanen and Aapala 2015) (II). In October 2014, the CMPP preparation was temporarily ceased in order to figure out options to implement CMPP voluntarily and in December 2014, CMPP was revised to a voluntary program (II). In October 2015, the working group preparing CMPP released its proposition for protection covering 117 000 ha.

2.4.2 Study design

CMPP candidate mires included open and wooded mires, and small areas of mineral soil forests (II). In this study, only wooded mire habitat types occurring in the boreal zone, i.e. spruce and pine mires were included. In Finland, both are commonly in a forestry use.

Four groups of mires were composed (II). The experimental group was composed of the wooded mires with the candidate status and the control group of the wooded mires without the status. Experimental and control groups were divided into spruce mires and pine mires. Their harvesting rates were analyzed during five years (2013–2017) after letters notifying candidate mires' landowners of their lands' conservation potential were sent. As a response variable for the overall harvesting rates over the five years, logged and unlogged area (hectares) within each of the groups were calculated.

Candidate mires were mostly in a natural state or close to it (II). If they had been highly modified or degraded by human, they would not have hosted typical mire biodiversity features, nor been chosen as the potential sites to CMPP. Due to the desire to protect candidate mires as hydrological entities, some of them enclosed small degraded parts which were planned to be restored after the protection. However, the age and timber volume of the candidate wooded mires likely represent those of older forests. To make the experimental and control groups to be equivalent, only the candidate and non-candidate wooded mires belonging to the two most mature forest development class (advanced thinning stands and mature stands) were included to the analyses. Additionally, the average diameters of tree stands in logged experimental and control groups were calculated. They did not differ remarkably indicating that the timber quality was similar independent of the conservation potential (Table 1).

TABLE 1 Average diameters of tree stands in logged experimental (mires with the candidate status) and control groups (mires without the candidate status).

Habitat type	Group	Area (ha)*	Avg. diameter (cm)	SD
Spruce mires	Experimental	86	18.11	4.91
	Control	39 264	18.69	5.16
Pine mires	Experimental	493	17.12	4.49
	Control	69 480	17.73	5.95

*Areas differ from the ones used in the study's statistical analyses as stands lacking the average diameter were removed before calculating the average diameters for the total areas.

Study period was set to be January 2013–December 2017 (II). Since CMPP was publicly informed from January 2013 onwards, it was not reasonable to study the harvesting rates earlier as if the candidate wooded mires had been logged earlier, before preparing CMPP, they would not have been selected as the candidate mires in the first place.

In Finland, the Forest Act obligates forest owners to make a notification of forest use before logging of their forest (II). Practically, all loggings are executed after submitting a notification since an industrial agent such as a timber buyer or a logging planner commonly makes the notification. Illegal loggings are very rare which is verified by a well-working law enforcement (Finnish Forest Centre 2018). Therefore, notifications were applied as surrogates for the loggings (II).

2.4.3 Data

For the analyses, eight different spatial data were combined: unlogged and logged spruce and pine mires with the candidate status, and unlogged and logged spruce and pine mires without the status (Table 2) (II).

To compile the data of all wooded mires in Finland, publicly available spatial forest resource data including detailed information of Finnish forests was applied (https://urly.fi/1jgz) (II). The data covers the majority of privately owned forest land but mostly it does not include government- and municipalities-owned lands. However, as majority of forest land in Finland is privately owned (Official Statistics of Finland 2011–2016), the data coverage can be considered to be representative (II). To make the data of all wooded mires to cover only advanced thinning or mature spruce and pine mire stands, the forest resource data was outlined according to the habitat type and forest development class (II).

To compile the data of all logged wooded mires in Finland, publicly available spatial data of the forest use notifications including information of logged forest stands was applied (https://urly.fi/1jgF) (II). The data served as a source for all the wooded mire stands that were advanced thinning or mature ones and logged in 2013–2017. However, many of the notifications lacked the information of a habitat type since it is not an obligatory field in the notification. To complete the information about the habitat type, the notification data was joined with the above mentioned data of all wooded mires and set the latter to act as a primary source for a habitat type. However, the data of all wooded mires did not cover all the stands in the forest use notification data. If the notifications on these stands included the information of a habitat type, it was used as the habitat type information. If the habitat type information was not available in either of the data, the stand in question was excluded from the analysis.

The data of all logged wooded mires was detached from the data of all wooded mires producing four data: unlogged and logged spruce and pine mires covering whole Finland (II).

The final eight data of unlogged and logged non-candidate and candidate spruce and pine mires were compiled by detaching the candidate spruce and pine mires from the above-mentioned data of all unlogged and logged wooded mires (II). This was made by means of a separate data that covered locations of the CMPP candidate mires (Alanen and Aapala 2015) (I, III).

TABLE 2 Sample sizes of the final processed data.

Group	Habitat type	Logging status	Hectares	No. of notifications
Experimental (mires with the candidate status)	Spruce mires	Unlogged	2 198	NA
		Logged	183	235
		Total	**2 381**	**235**
	Pine mires	Unlogged	6 661	NA
		Logged	981	700
		Total	**7 642**	**700**
Control (mires without the candidate status)	Spruce mires	Unlogged	357 415	NA
		Logged	78 196	54 314
		Total	**435 611**	**54 314**
	Pine mires	Unlogged	599 896	NA
		Logged	136 390	61 473
		Total	**736 286**	**61 473**

2.4.4 Final data processing

Assembling the datasets caused multiple fragment stands that were too small to be real forest stands (II). Size distributions of the forest stand fragments were analyzed separately for all eight datasets and accordingly estimated, that excluding stands ≤ 0.14 ha would reduce the number of artificial stands without eliminating many of the real small stands. It is likely that all of the artificial stands were not excluded and likewise, some of the existing small stands were excluded.

However, there were not reasons to expect any bias in the data caused by the exclusion so the data was considered to be reliable. All the data was processed with ArcMap 10.

2.4.5 Statistical analysis

Harvesting rates per 5 years were analyzed on mires with and without the candidate status and separately for pine and spruce mires against randomized harvesting rate distributions (II). To create the distributions, the total habitat-specific logged hectares of all pine and spruce mires were set randomly to locate on the whole area of the respective habitat. Randomization was performed with RStudio version 1.1.456 and replicated 1 000 times. Replicates were compiled into a distribution describing how large proportion of the logged hectares would randomly locate on the candidate mires of each habitat type.

2.5 Ecological, economic, and social trade-offs in mire protection

2.5.1 Data

Biodiversity data and the data indicating mires' ecological condition were same as in chapter I. In addition to them, a cost data layer and a data layer of landowners' resistance to protection were applied.

Cost data layer was based on each mire's economic compensation value, including the value of land area, tree-stand, and an administrative cost (III).

Data layer of landowners' resistance to protection was based on a landowner survey and on negotiations with forestry and peat mining companies both implemented by the Finnish Ministry of the Environment (III). Survey was sent to the citizen landowners of the candidate mires that were proposed for protection. One of the questions asked was: "Would you consider protecting your property shown in an attached map for the market price?" Answer options were "yes", "I don't know", and "no". All "no" answers were considered as a resistance to protection. The "I don't know" answers were interpreted as positive ones as unsure landowners could likely be persuaded to protect. The survey was not sent to all citizen landowners of all candidate mires and not all survey recipients answered, so applying only the observed preferences of landowners would have restricted the analysis area impractical small for prioritization. Therefore, citizen landowners' resistance to protection was extrapolated to cover all the candidate mires owned by them. As landowners' attitude towards conservation can depend on their relationship to regional environmental authorities (Salomaa *et al.* 2016), extrapolations were made independently for 10 provinces, based on their observed distributions of resistance (III). In each province, the resistance distribution was calculated from the observed mire-specific percentual resistance (III). According to the distribution, the resistance percentages were then randomized to the mires lacking observed resistance.

Three provinces were totally excluded from the survey, so for candidate mires located on them, the percentual average resistances were randomized from the distribution of all 10 provinces included to the survey. Finally, the resistance layer included the observed resistance of citizen-, government-, and company-owned candidate mires and the extrapolated resistance of those citizen-owned candidate mires who were not covered by the observed resistance data.

All data layers were converted to 50 x 50 m raster data (III).

2.5.2 Spatial prioritizations

Prioritizations were made with Zonation software 4.0.0 (Moilanen *et al.* 2005, Moilanen *et al.* 2014) (III). The base model for the analysis was the same as in chapter I. Here, a cost layer and a layer about landowners' resistance to protection of their land were added. The cost layer de-emphasized high-priced mires (III). In order to maintain hydrological connectedness of mires, they were prioritized as planning units, defined by mire experts (I).

Three prioritization scenarios were designed (III). They were different in how they considered landowners' resistance to protection. In *Voluntary* scenario, all candidate mires (i.e. planning units) where at least one owner, either a citizen or a company, resisted protection were excluded from protection regardless of their average representation of biodiversity features or financial costs. In *Obligatory* scenario, mires with high feature representation but low costs were aimed, ignoring landowners' resistance to protection. In *Balancing* scenario, resistance was considered as a continuous variable, i.e. proportion of each candidate mire's landowners resisting protection. Negative weight was given to resistance in order to de-emphasize resisted mires in prioritization. To determine a suitable level of negative weight for resistance, the analysis was iterated with several differing levels of weighting and investigated the related trade-offs. We iterated the analysis with weights of 0, –10, –20, –50 and –100 for the resistance to investigate the effect of the weight and resulting trade-offs. Performance curves for resistance showed that using the weight of –100 would have been almost equivalent to the total exclusion of all resisted mires (Fig. 4) (III). Instead, the weight of –50 effectively excluded resisted mires from the top fraction, still allowing valuable biodiversity features located in resisted mires to prevent all the exclusions (III). On the other hand, biodiversity representation produced by the weight of –20 differed only little from that produced by the weight of –50, but caused almost double resistance. Therefore, we chose the weight of –50 to serve as a minus weight for the resistance layer in *Balancing* scenario. In summary, *Balancing* scenario aimed simultaneously to avoid resisted mires and to emphasize high feature representation and low costs.

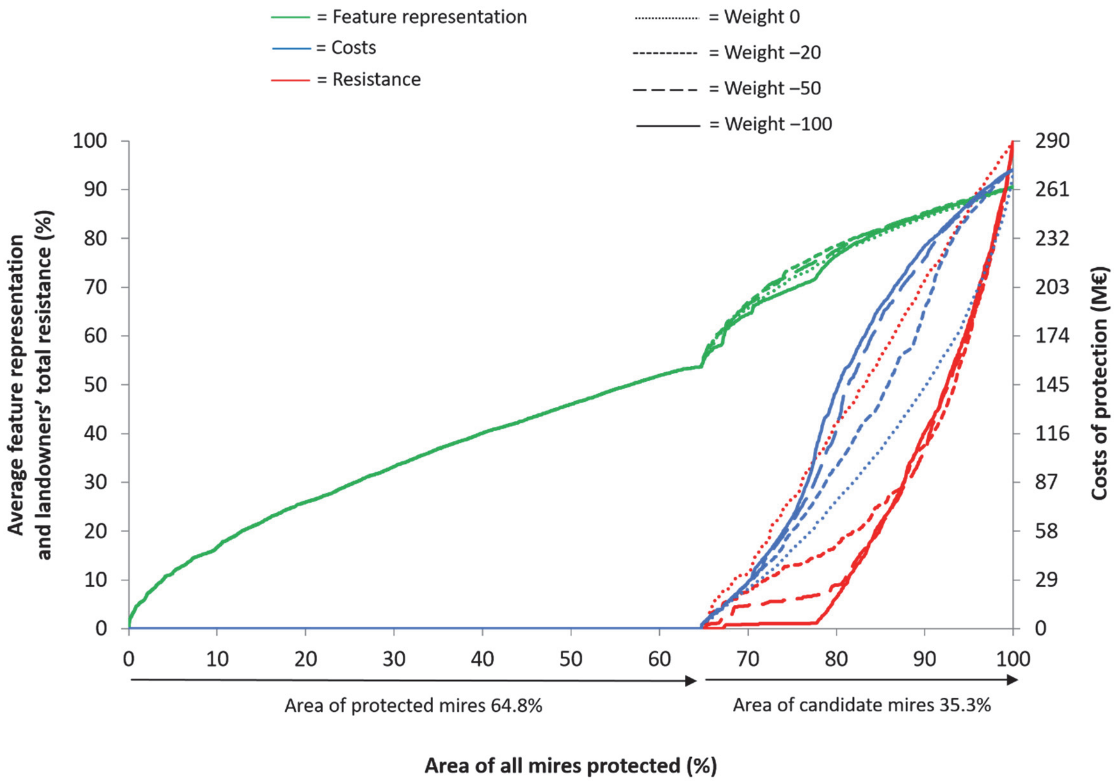


FIGURE 4 Average representation of biodiversity features (green curve), landowners' total resistance (red curve), and costs of conservation (blue curve) for each prioritization testing the effect of different minus weights of resistance to protection. Small dotted lines represent zero weighting, medium dotted lines weight of –20, dashed lines weight of –50, and solid lines weight of –100. Adapted from chapter III.

2.5.3 Comparison between scenarios

Scenarios were compared by observing their levels of average biodiversity feature representation protected (% of total feature representation included in prioritizations), costs (million euros), and resistance (% of all landowners) (III). To make comparisons comprehensible, *Voluntary* scenario was chosen as a reference for two other scenarios because it explicitly defines the candidate mires that are completely free from landowners' resistance to protection. *Obligatory* and *Balancing* scenarios' average feature representation, costs, and total protected area were compared against those set by *Voluntary* scenario.

3 RESULTS

3.1 Preparing the complementary mire protection

3.1.1 Expansion of CMPP

The working group assigned for protection approximately 117 000 ha of candidate mires (Alanen and Aapala 2015) (I). The ecological value of the network of protected mires increased as a function of gradual addition of candidate mires, as shown by the performance curves (Fig. 5) (I). The highest ranked 5% of the analysis area, corresponding to an approximately 8% increase of protected mire area or 1/3 of all candidate mires chosen for protection, increased the coverage of protected biodiversity feature representation on average by 39%. High cost-effectiveness was primarily achieved via area additions for narrow-range features that have missing or low representation in the existing protected area network. Coverage of the most threatened mire complexes and habitats improved significantly more than coverage of all features, showing a 68% relative improvement for the 8% area increase.

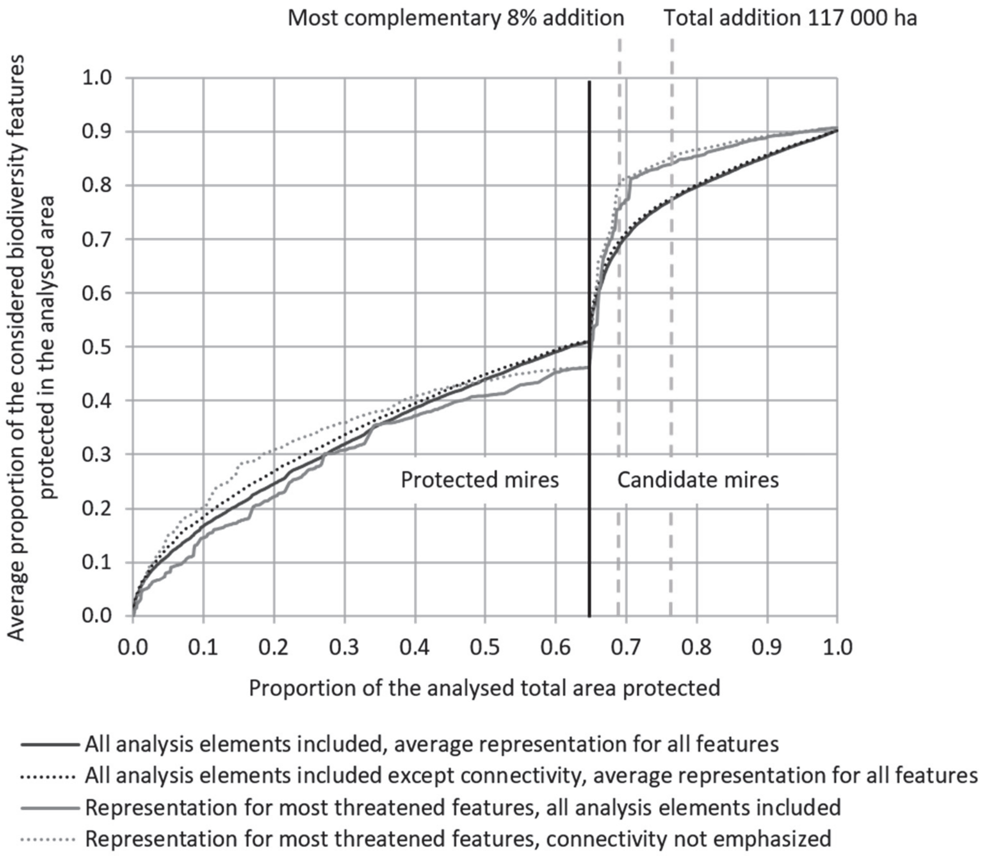


FIGURE 5 Average representation of biodiversity features for different versions of the analysis. Performance curves describe how the coverage of feature representation changes as a function of area added into protection. The black vertical line indicates division between the existing mire conservation network (left from the line) and candidate mires (right). The steep rise of the curves to the right of the black line means that some species or habitats are missing or poorly represented within the existing protected mires (see also Fig. 6), but that the coverage of these features can be improved rapidly with additional mires, until an increase of representativeness starts to saturate. The left vertical dashed gray line shows the amount suggested to be chosen according to the prioritization analysis to ensure a complementary solution (8% area addition to what is already protected), and the right vertical dashed gray line marks the total additional area suggested to be protected by CMPP. Adapted from chapter I, i.e. Kareksela *et al.* 2020.

The prioritization analysis also improved the balance between the protected biodiversity features (I). It effectively raised the representation levels of the features with lowest representations at the current mire conservation network (Fig. 6) (I).

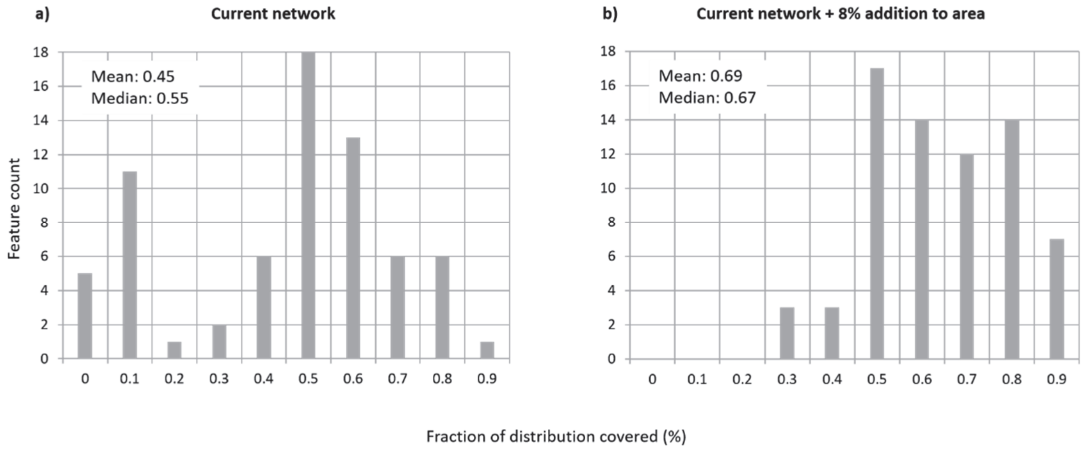


FIGURE 6 Histogram of coverage of biodiversity features relative to the features' abundances. Small water bodies and modelled distributions of birds are excluded. Comparison of the histograms for the current network (a) and with 8 % increase in its area (b), demonstrate the filling in the biodiversity gaps, i.e. improving the representation of the features least well represented in the current network. The x-axis shows the fraction of a features' analyzed abundance protected currently (a) and in a +8% situation (b), and the y-axis shows the number of features that have that fraction of its abundance protected. Note that the +8% represents the mires chosen for protection according to the prioritization analysis, i.e. 1/3 of the total area chosen for protection. Thus, the final increase in total biodiversity feature representation is higher than shown here. Adapted from chapter I, i.e. Kareksela *et al.* 2020.

Performance curves were also used to investigate a potential trade-off between biodiversity coverage and spatial connectedness (Fig. 5) (I). Inclusion of connectivity did not have a significant negative effect on the biodiversity feature representation although it did have a small downward effect for the most threatened features (Fig. 5) (I).

Twenty-nine percent of the total candidate mires' area required restoration and on average 55% of these were on mires that host one or more highest-weighted mire complexes and/or habitats (I). Of the 8% area addition chosen for protection according to the prioritization analysis, 20% required restoration. Of these, 78% located on mires that host the highest-weighted mire complexes and/or habitats. This means that the analysis effectively chose areas in good condition, i.e. areas with lower need of restoration. However, the still remaining need for restoration was strongly associated to the mires with high-priority habitats, directing potential future restoration efforts to the mires where restoration would most benefit biodiversity.

3.1.2 Integrating analysis results into decision-making

The working group decided the final set of mires that would be protected to complement the existing mire protection network (I). The prioritization experts recommended the number and identity of mires that should preferably be chosen according to the prioritization analysis to retain its greatest benefits. This was the 8% area addition, or approximately 1/3 of the area that could be chosen within the CMPP's area-based target limitation (Fig. 5) (I). The remaining 2/3 of the targeted additional area for protection were then chosen according to the highest scoring points (I). This approach was welcomed and strongly supported by the working group, leading to a successful integration of the prioritization results into the process of deciding which mires should be protected.

3.2 (Non)existence of pre-emptive loggings on wooded mires

7.7% (183 ha) of spruce mires and 12.8% (981 ha) of pine mires with the candidate status were logged based on hectares covered with submitted forest use notifications (II). Respective numbers for spruce mires without the candidate status were 18.0% (78 916 ha) and for pine mires 18.5% (136 390 ha). The candidate mires were logged significantly less than the mires without the candidate status (Fig. 7) (II). The average annual harvesting rates of the candidate spruce and pine mires were 1.54% and 2.57%, respectively (II).

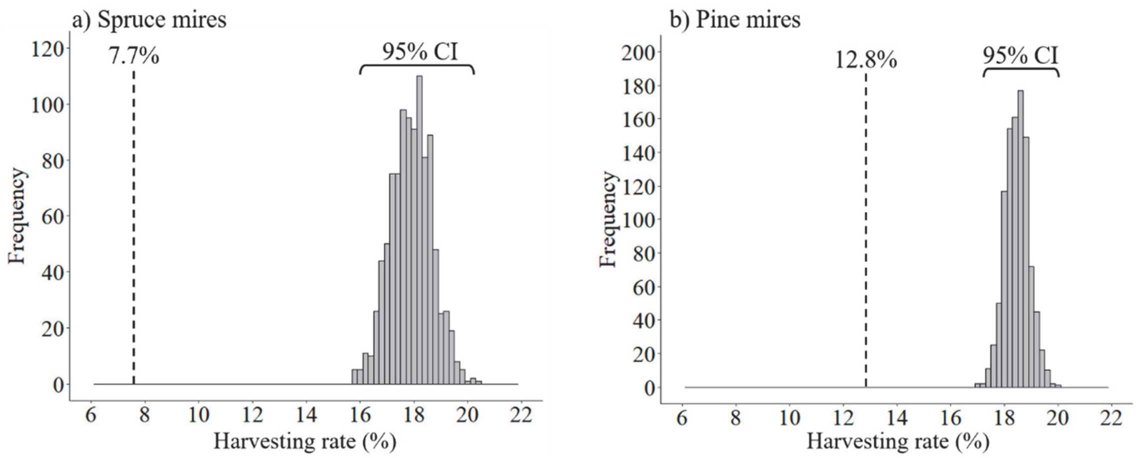


FIGURE 7 Grey bars show how large proportion of logged hectares would randomly locate on the candidate a) spruce mires and b) pine mires, when randomization is replicated 1 000 times. Actual harvesting rates per 5 years of the candidate mires are marked with dashed vertical lines. Adapted from chapter II.

Within the years, both the wooded mires with and without the candidate status had a seasonal variation in the number of submitted forest use notifications (Fig. 8) (II). Notifications were regularly submitted more in the autumn and winter, and less in the spring and summer. On the candidate spruce mires, the highest numbers of notifications during the study period were submitted in October 2014 (11 notifications), in April 2013, and in January 2016 (9 notifications during both). Respective months and years for the candidate pine mires were October 2017 (25 notifications), and October and November 2014 (24 notifications during both). Taking into account the seasonal variation, there were no obvious peaks in the numbers of submitted notifications in May–July 2013, when landowners were notified that their mires are candidate mires for CMPP, nor in October–November 2014, when the option of land expropriations was rejected.

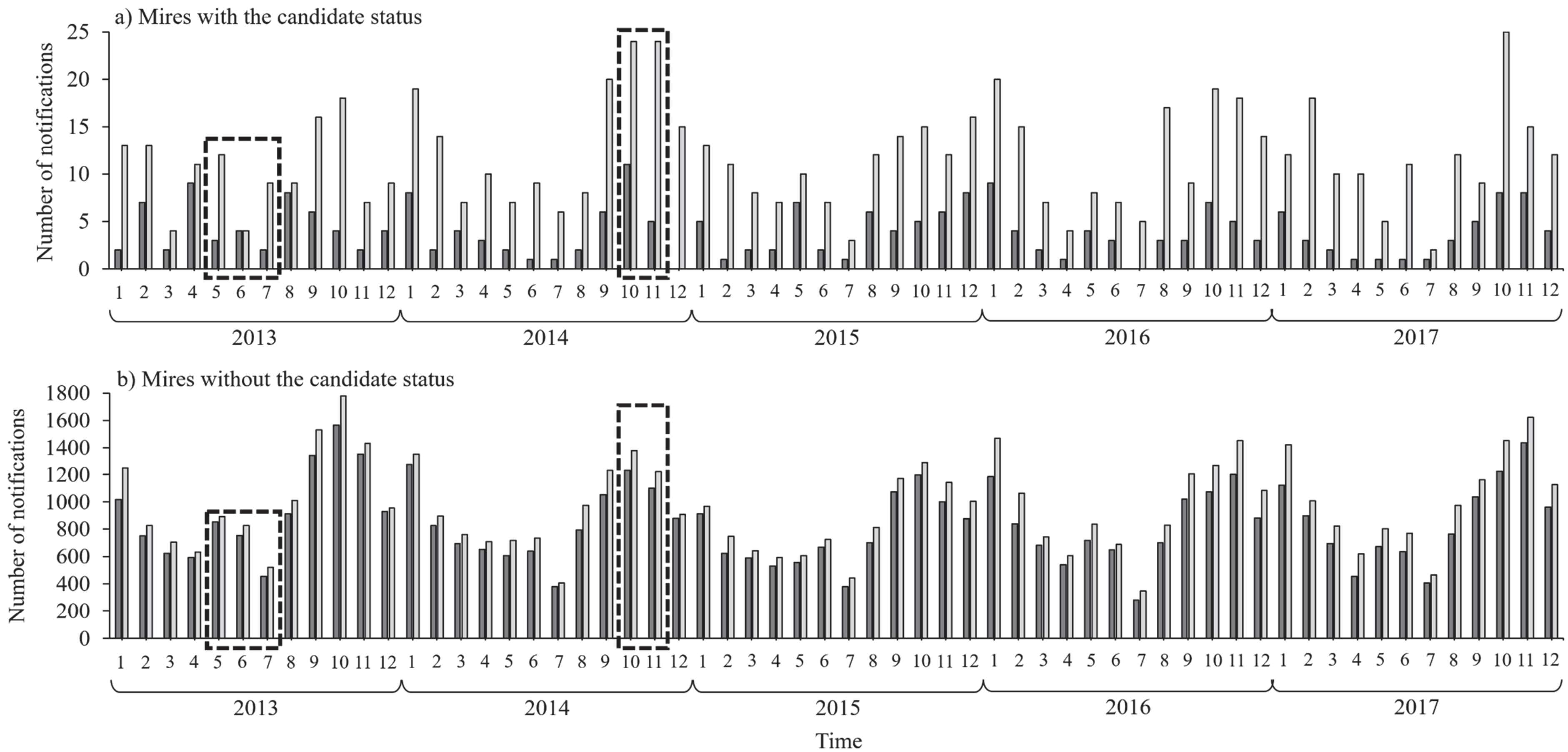


FIGURE 8 Number of forest use notifications submitted per month in 2013–2017. Dark gray bars represent spruce mires and light grey bars pine mires. a) The mires with the candidate status. b) The mires without the candidate status. The boxes with dashed lines represent the months when we expected the number of notifications to rise due to the certain events concerning CMPP. Adapted from chapter II.

3.3 Ecological, economic, and social trade-offs in mire protection

Of the total area included in the prioritizations, the existing protected mires covered 64.8% (601 700 ha) and the candidate mires 35.2% (327 300 ha) (Fig. 9) (III). 53.7% of average representation of biodiversity features included in the prioritizations were situated on the existing mire conservation network (Fig. 9, Table 3) (III).

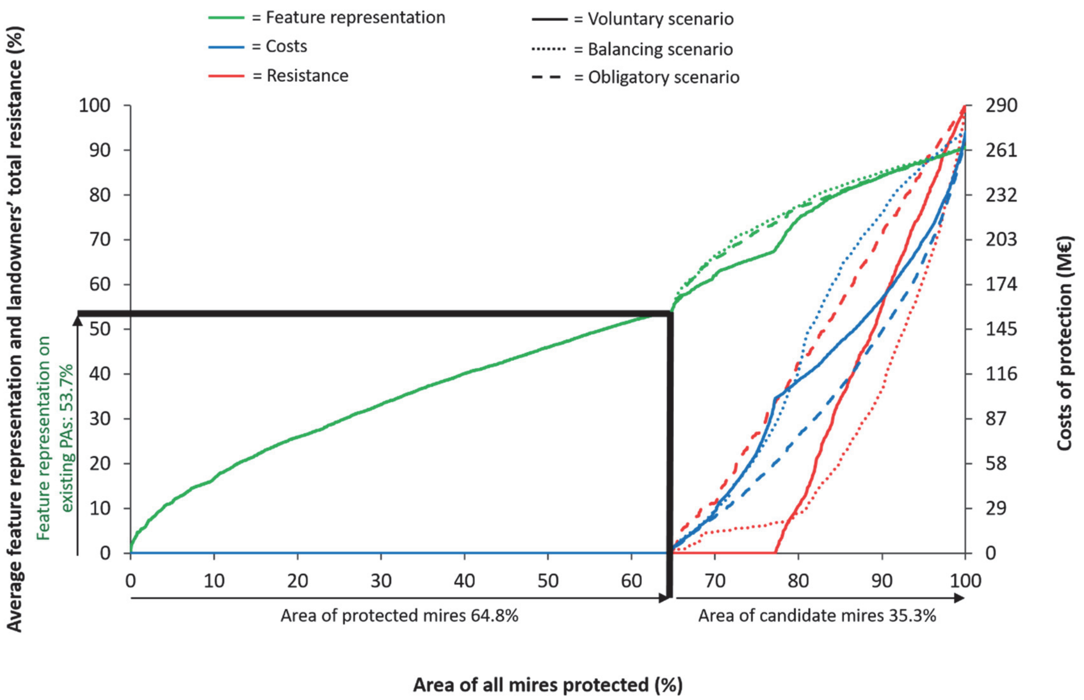


FIGURE 9 Thick black lines represent the average representation of biodiversity features (53.7%) and area (64.8%) covered by the existing mire conservation network. Green curves represent average biodiversity, red curves landowners' total resistance to protection, and blue curves conservation costs. Solid colored lines represent *Voluntary* scenario, colored dotted lines *Balancing* scenario, and colored dashed lines *Obligatory* scenario. Average feature representation does not reach 100% because the included mire complexes and habitats suffer from decreased condition caused by drainage and expressed in the curves. Adapted from chapter III.

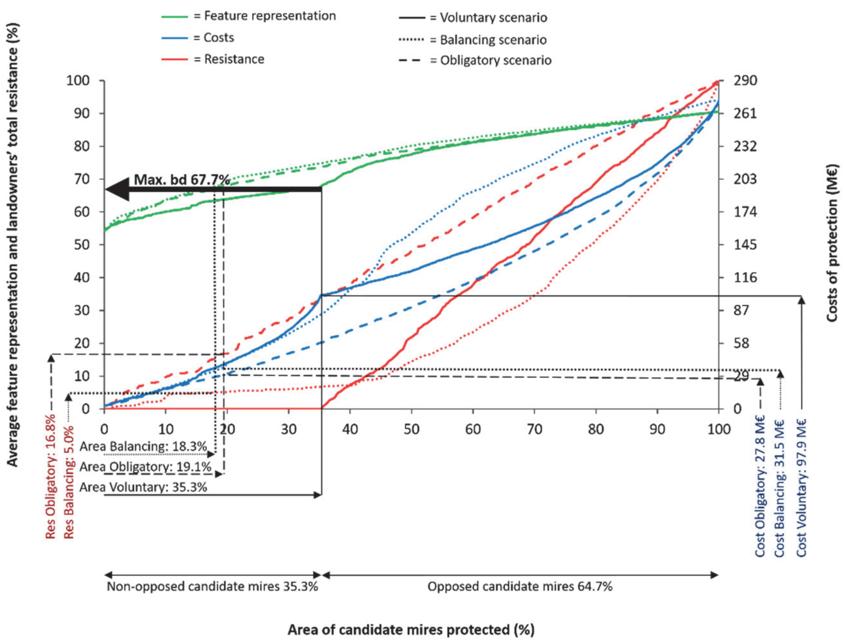
Differences between scenarios emerge from the fact that in each scenario, the ranking of mires is different and therefore, a different combination of mires is protected. *Voluntary* scenario, including all candidate mires completely free from landowners' resistance to protection, increased the average protected representation of biodiversity features by 26.1%, from the existing 53.7% to 67.7%, with the costs of 97.9 million euros (Table 3, Fig. 10) (III). Protected area increased 115 400 ha (19.2%), which is a bit less than the original proposal where landowners' preferences were ignored (117 000 ha) (Alanen & Aapala 2015) (I, III).

With the same average representation of biodiversity features as in *Voluntary* scenario, the area protected and the costs needed to protect it were much lower both in *Balancing* and *Obligatory* scenarios than in *Voluntary* scenario (III). Both *Balancing* and *Obligatory* scenarios were able to fulfill the average feature representation of *Voluntary* scenario by almost half the area (59 800 and 62 500 ha vs. 115 400, respectively) (Fig. 10a, Table 3) (III). Accordingly, the costs of protection decreased remarkably (III). The costs of *Balancing* scenario were 66.4 million euros lower and those of *Obligatory* scenario 70.1 million euros lower than in *Voluntary* scenario. *Balancing* scenario performed well in keeping the total landowners' resistance low at 5.0%, while in *Obligatory* scenario the total resistance was 16.8%.

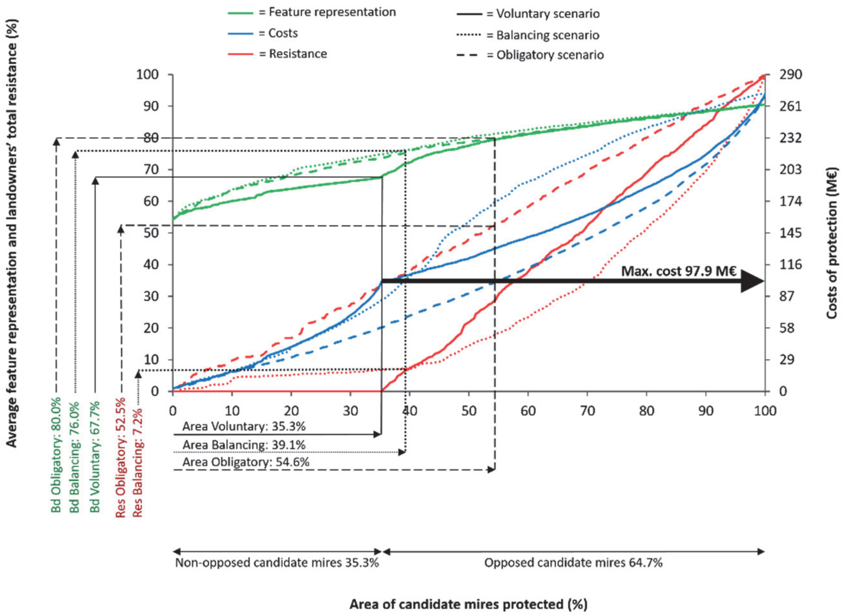
With the same budget for protection as in *Voluntary* scenario, *Balancing* scenario resulted in 10.5% larger total area for protection with 12.3% higher average feature representation than *Voluntary* scenario (Fig. 10b, Table 3) (III). Total resistance still stayed relatively low at 7.2% (III). In *Obligatory* scenario, total area protected was 54.8% larger and the average feature representation 18.2% higher than in *Voluntary* scenario. Area protected covered already over half of the candidate mires (176 600 ha) and the total resistance increased to 52.5%.

With the same total area protected as in *Voluntary* scenario (115 400 ha), *Balancing* scenario achieved 10.5% higher average feature representation with 17.1 million euros lower costs than *Voluntary* scenario (Fig. 10c, Table 3) (III). *Balancing* scenario performed rather well also in minimizing the total resistance as it remained relatively low at 6.7% (III). Relative to *Voluntary* scenario, *Obligatory* scenario covered 9.0% more average feature representation with 42.1 million euros lower costs. However, the total resistance increased to 33.7%.

a)



b)



c)

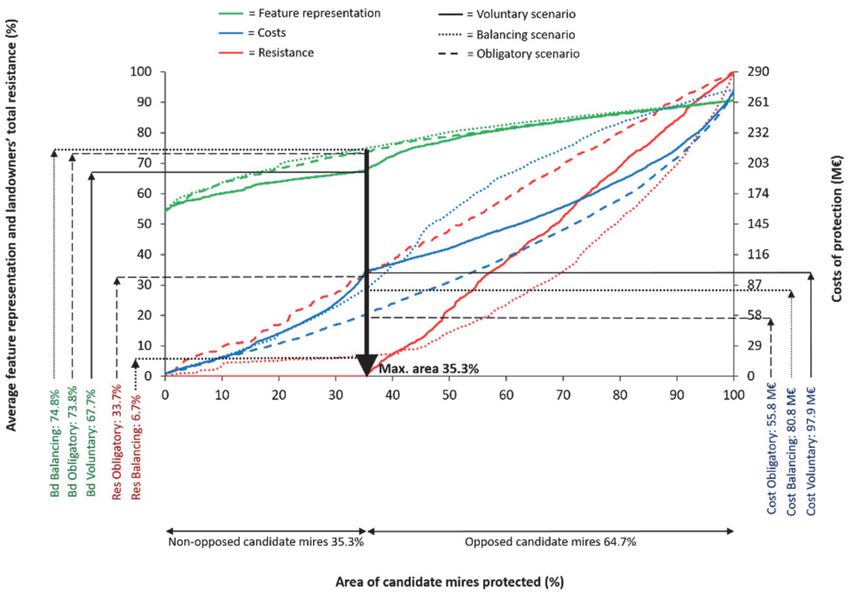


FIGURE 10 Average representation of biodiversity features (%) (green curves), landowners' resistance to protection (%) (red curves), and costs of protection (million euros) (blue curves) for each conservation scenario. Thick black arrows point out the values produced by *Voluntary* scenario against which other solutions are compared to. Black solid lines and arrows represent *Voluntary* scenario, black dotted lines and arrows *Balancing* scenario, and black dashed lines and arrows *Obligatory* scenario. a) Feature representation protected is that of *Voluntary* scenario, i.e. 67.7%. b) Costs for protection are those of *Voluntary* scenario, i.e. 97.9 million euros. c) Area for protection is that of *Voluntary* scenario, i.e. 35.3% of the candidate mires' area. Average feature representation does not reach 100% because the included mire complexes and habitats suffer from decreased condition caused by drainage and expressed in the curves. Adapted from chapter III.

TABLE 3 Average representation of biodiversity features, financial costs, and area of existing protected mires and *Voluntary*, *Balancing*, and *Obligatory* scenarios when average feature representation, costs, or area has been fixed according to the numbers produced by *Voluntary* scenario. Adapted from chapter III.

	Average feature representation fixed				**Costs fixed**				**Area fixed**			
	[a]Biodiversity %	[b]Cost M€	[a]Area % (ha)	[b]Resistance %	[a]Biodiversity %	[b]Cost M€	[a]Area % (ha)	[b]Resistance %	[a]Biodiversity %	[b]Cost M€	[a]Area % (ha)	[b]Resistance %
Existing protected mires	53.7 | NA	NA	64.8 | NA (601 700 | NA)	NA	53.7 | NA	NA	64.8 | NA (601 700 | NA)	NA	53.7 | NA	NA	64.8 | NA (601 700 | NA)	NA
Voluntary scenario	67.7 | 37.2	97.9	77.2 | 35.3 (717 100 | 115 400)	0	67.7 | 37.2	97.9	77.2 | 35.3 (717 100 | 115 400)	0	67.7 | 37.2	97.9	77.2 | 35.3 (717 100 | 115 400)	0
Balancing scenario	[c]67.8 | 37.3	31.5	71.2 | 18.3 (661 500 | 59 800)	5.0	76.0 | 60.1	[c]97.8	78.5 | 39.1 (729 200 | 127 500)	7.2	74.8 | 56.7	80.8	77.2 | 35.3 (717 100 | 115 400)	6.7
Obligatory scenario	67.7 | 37.2	27.8	71.5 | 19.1 (664 200 | 62 500)	16.8	80.0 | 70.8	[c]97.8	84.0 | 54.6 (780 300 | 178 600)	52.5	73.8 | 53.7	55.8	[c]77.1 | 35.1 [c](716 400 | 114 700)	33.7

[a]The first number represents %-values (hectares in brackets) after protection (existing protected mires + selected candidate mires), while the second number after the vertical line shows the %-values (hectares in brackets) of the candidate mires that were included into the scenarios.
[b]The numbers represent values calculated for the candidate mires that were included into the scenarios.
[c]As the prioritization was conducted in planning units, the area or costs cannot be fixed exactly the same in each scenario (i.e. cut in the middle of a planning unit). This results in small variation to the fixed numbers.

4 DISCUSSION

4.1 Well planned is only half done

Site selection for CMPP produced an ecologically successful conservation solution. Additionally, it was a decision-making process successfully utilizing scientific and practical knowledge. Spatial prioritization analysis played a crucial role both in producing the conservation solution and in combining scientific and practical knowledge.

The prioritization analysis was able to find the candidate mires where protection would most efficiently fill the gaps in the representation of biodiversity features on the existing mire conservation network. It also found the candidate mires where high conservation value outweighed the lowered condition that was caused by partial ditching. Restoring drained parts of these mires would safeguard present occurrences of a large amount of biodiversity features, which could then provide a species source for restored parts. Hence, the analysis also identified mires, whose restoration would be highly beneficial for biodiversity.

Systematic analysis enabled observing differences in analysis results that may arise because of considering connectivity of mires. Differences were small between the analysis versions with and without connectivity, so the working group agreed that considering connectivity would be beneficial for biodiversity in long term regardless of the fact that the representation of the most threatened features in the conservation solution would be a bit lower than without considering connectivity (see also Crooks and Sanjayan 2006).

Knowledge provided by the working group had a significant role in making the prioritization analysis to produce ecologically successful results. Building the analysis together with the prioritization experts and the working group made the analysis results to be relevant with respect to the working group's objectives. Co-production also enabled the use of the working group's expert knowledge in a more analytical way (Drescher *et al.* 2013). The working group was satisfied with clear visualization of the analysis results, which enabled people less involved in prioritization to make well-informed decisions about mire protection. The group was able to consider how much to follow the analysis results and how much to select mires based on the scoring method and expert knowledge without losing the ecological complementarity of the conservation solution. All parties felt that the multidirectional knowledge transfer resulting from the analysis co-production had a significant impact on the practical use of the prioritization results, benefiting the stakeholders in the working group, the prioritization experts, and mire conservation itself (see also Bertuol-Garcia *et al.* 2018). Hence, the prioritization analysis applied together with the scoring method fulfilled the characteristics of the Structured Decision Making: the process of selecting mires for CMPP was supported by a valid technical analysis, requiring a clear value judgement between different ecological considerations, and engaging stakeholders in good decision-making practices (Gregory *et al.* 2012). Here, the science-practice gap seemed to be efficiently fulfilled in a sense that the result of scientific prioritization method was disseminated into a practical conservation solution yet finally, the solution was not implemented due to unpredictable political decisions (III).

One could ask why 2/3 of the area chosen for protection was selected primarily according to the scoring method, because compared to systematic spatial prioritization methods, scoring is less efficient and does not fulfil the complementarity criteria (Pressey and Nicholls 1989, Moilanen *et al.* 2009b). Despite their weaknesses, scoring methods are tempting as they are often seen easier to understand and implement than spatial prioritization methods, which require expertise to be correctly utilized and interpreted. It should also be noted that although prioritization methods are scientifically more rigorous and consider complementarity, neither the prioritization nor the scoring approaches are free from value judgements. Both Zonation and the applied scoring approach require weighting of certain things based on expert opinions (Moilanen *et al.* 2014, Alanen and Aapala 2015). Additionally, successfulness of results produced by spatial prioritization methods are dependent on the quality of input data (Lehtomäki and Moilanen 2013). Mire experts could have had knowledge that was not included in input data or analysis settings. Possibly, they were able to inspect single mires more precisely than the prioritization analysis, although they did not perceive the big picture as well as the analysis. Therefore, it is possible that the prioritization method and the scoring approach complemented each other. Either way, the described procedure combining both the methods was welcomed by the working group. Additionally, 1/3 of the mires chosen for protection according to the prioritization analysis filled the needs of biodiversity complementarity very well meaning that the rest could have chosen according to the scoring method without leaving irreplaceable biodiversity features outside protection. Hence, the protected biodiversity gains likely would not have differed much, be the selection made purely according to prioritization, or according to prioritization and scoring.

The described process seems to give a relatively good answer to the question “how to protect nature”. But due to a sudden political turmoil leading to replacement of the Minister of the Environment and changing CMPP as a voluntary initiative (Salomaa *et al.* 2018) (III), the carefully made and satisfying CMPP plan never fully converted into practical conservation actions. Many papers have presented methods to evaluate and prepare for risks that could endanger implementation or target achievement of conservation plans, but often they have focused on threats that may threaten biodiversity the conservation plans aim to safeguard (Tulloch *et al.* 2015). Therefore, they could not have helped in the case of CMPP. (A threat on biodiversity was taken into account also here: a ditching level was included in the prioritization analysis to avoid protecting mires that are in danger of drying and losing their characteristic features.) In hindsight, the working group could have principally planned CMPP so that landowners' opinions would have affected site selection. However, since there were the decision made by the government to enable land expropriations if needed, and since CMPP was intended to be resourced, planned, and implemented during the 4-year government term, it is understandable that the working group did not make extra work and allocate extra resources to figure out landowners' willingness to protect. The political changes concerning the premise of CMPP implementation were truly unpredictable since the same government, which had made the decision to allow land expropriations in CMPP, appointed the minister that rejected the option of expropriations, without rest of the government preventing the rejection. In addition to fluctuating political preferences, also other societal factors such as complex decision-making processes (Kareksela *et al.* 2018) can prevent implementation of conservation plans or prioritizations, no matter how well they were prepared.

4.2 Why Finns did not log pre-emptively

Unlike rumors and fears implied, CMPP candidate wooded mires were not systematically pre-emptively logged. The lack of systematic pre-emptive behavior contradicts with evidence collected from USA and Australia (Brook *et al.* 2003, Lueck and Michael 2003, Zhang 2004, Simmons *et al.* 2018b). The reasons behind the lower than average harvesting rates on candidate wooded mires can be many and they can just be speculated as exact knowledge does not exist, but it is likely that the landowners' behavior is affected by the society they live in. Generally, the Finns seem to react positively to conservation, since majority of citizens, regardless of their socioeconomic or demographic status, agree that protection of mire habitats and species is important (Tolvanen *et al.* 2013). Amongst owners of CMPP candidate mires, there were more persons having a positive attitude toward protection of their mires than the ones having a negative attitude (Alanen and Aapala 2015). This is encouraging given that past involuntary conservation initiatives such as Natura 2000 and the Conservation Programs for Old Growth Forests and for Shores caused heavy conflicts in Finland (Hiedanpää 2005, Paloniemi and Vilja 2009). Likely, the first voluntary forest conservation program METSO, started in the beginning of 2000s, have helped to overcome previous biodiversity conflicts (Paloniemi and Vilja 2009), possibly making the public attitude receptive to new conservation initiatives. Furthermore, a fair compensation of the land can increase the likelihood of pro-conservation behavior and decrease the likelihood of pre-emptive behavior (e.g. Langpap 2006, Ferraro *et al.* 2007, Byl 2019). Therefore, it is possible that the Finnish compensation practice to pay market price for protected sites satisfy landowners, unlike different compensation practices in some other countries. For instance, principles of compensation practices concerning The Endangered Species Act in USA are fundamentally different compared to those in Finland (Donahue 2005).

Despite the low rate, some of candidate wooded mires were still harvested. The seasonal variation in their harvesting rates imitating that of non-candidate wooded mires and the lack of obvious logging peaks after sending notification letters implicate, however, that instead of pre-emptive behavior, there may have been some other reasons to log. Landowners may have simply followed their long-term logging plans that are often made in cooperation with a local forestry specialist. There is evidence that forestry-oriented landowners trust forestry specialists and prefer cooperating with them also in conservation issues rather than with environmental officers (Paloniemi *et al.* 2006). Therefore, forestry-oriented landowners may have actively disregarded the information provided by environmental authorities about the high conservation value their mires had. Such behavior may be expected particularly if a landowner's income is dependent on the actualized loggings. Nevertheless, there is earlier evidence of Finnish forest owners intentionally taking actions to harm flying squirrel (*Pteromys volans*) (Jokinen *et al.* 2018), the species which is listed in the Habitats Directive of the European Union and is therefore under strict protection (Council Directive 92/43/EEC). Since intentional harming of biodiversity is not unprecedented in Finland, it is possible that also some owners of candidate wooded mires executed intentional pre-emptive loggings.

Although biodiversity-rich candidate wooded mires were logged less than their non-candidate counterparts, even the low harvesting rates are problematic. If loggings on candidate wooded mires continued with the observed average annual rate of 36.6 ha (1.54%) for spruce mires and 196.2 ha (2.57%) for pine mires, it would take only 26 and 13 years to lose half of them, respectively. Furthermore, the working group preparing CMPP included 117 000 ha of the most ecologically valuable mires into the protection proposal, while 210 300 ha were excluded (Alanen and Aapala 2015) (I). After revising CMPP as a voluntary program, the excluded mires could serve as compensatory sites for the most ecologically valuable mires that would not be set aside due to some landowners' unwillingness to protect. Therefore, the likelihood to reach an ecologically representative mire conservation network decreases as increasingly larger area of candidate wooded mires are exposed to loggings, whether they were originally chosen for the most valuable ones or not.

Since many owners of candidate mires are conservation-minded (Alanen and Aapala 2015), it might be possible that at some point the loggings would cease even without protection. However, random factors such as transferring land property to the next generation or sudden acute need of money may lead to harvesting of a biodiversity-rich but non-protected mire even if the owner would have for now decided to set the mire aside by her/his own decision. Excluding biodiversity-rich forests from official protection is a potential threat for long-term persistence of biodiversity since forestry in Finland is so intensive that majority of forest sites are logged once they reach maturity (Natural Resources Institute Finland 2019).

There is a possibility that some other factors than the ones being explored here have had an impact on the harvesting rates on candidate wooded mires. The one worth to discuss is the possibility of candidate wooded mires being on average smaller-sized than non-candidate ones and therefore, possibly less prone to forestry practices. During heavy ditching campaign in 1960s and 1970s (Vasander 2006), large mires having a high potentiality for wood production or peat mining could have been more likely ditched than small ones, increasing the likelihood of large mires to degrade and therefore, to be excluded from CMPP candidate mires. Small forest stands, by contrast, may be silviculturally less attractive than larger ones as their share of logistical costs in timber revenue is greater. On the other hand, Finnish Forestry Management Associations often endeavor to centralize loggings to certain areas so that neighboring forest properties are logged at the same time, which lowers the logistical costs of small-sized regeneration-ready stands. This balances the effect of possibly smaller average size of candidate wooded mires on landowners' willingness to log. This remains as a speculation since areas of single non-candidate and candidate mires were not possible to calculate, because their borders were lost due to the data processing. However, even if there were some other reasons for candidate mires' low harvesting rates than landowners' awareness of their lands' conservation potential, the lack of obvious increases in the logging activity on candidate mires after notifying of their conservation value means that in any case, landowners did not engage in systematic pre-emptive loggings.

4.3 Unavoidable trade-offs and their alleviation

4.3.1 All scenarios have their pros and cons

When protecting nature, pure win-win situations are rare, if not non-existent (see also e.g. Guerrero *et al.* 2010, Lele *et al.* 2010, Knight *et al.* 2011, McShane *et al.* 2011, Adams *et al.* 2014). Comparison of *Voluntary*, *Obligatory*, and *Balancing* scenarios confirm that trade-offs between ecological, social, and economic considerations in private land protection cannot be fully avoided. Dimensions of the trade-offs between the level of average representation of protected biodiversity features, financial costs of conservation, and the amount of landowners' resistance to protection varied remarkably depending on how the landowners' resistance was considered in prioritization. The differences emerge due to different combinations of sites selected for protection.

A clearly positive aspect in *Voluntary* scenario was that all the protected mires were owned by conservation-minded owners, so none of land expropriations or any kind of persuading actions would need to be executed. However, cost-efficiency in *Voluntary* scenario was remarkably low. Technically, costs were minimized during prioritizing *Voluntary* scenario, but in practice they did not have an effect because to get close to the original proposal of protecting 117 000 ha (I), all the lands of conservation-minded owners needed to be included in the conservation solution. Low cost-efficiency is problematic as conservation resources are often scarce with respect to conservation aims and requirements (Geldmann *et al.* 2018), meaning that less money would be available for other conservation purposes and still, the biodiversity effect of the scenario is inferior compared to the other scenarios (see also Lewis *et al.* 2011). Perhaps surprisingly, *Voluntary* scenario also seems to represent somewhat unfair situation, since a single unwilling landowner among many willing ones shifts a whole mire to be out of reach for protection. This happens because protecting mires as unconnected fragments is not reasonable or cost-effective in long term, because their typical features are dependent on a water table level (Moore and Knowles 1989, Laine *et al.* 1995, Jungkunst and Fiedler 2007, Haapalehto *et al.* 2011, Maanavilja *et al.* 2014). Accordingly, mires need to be protected as hydrological entities. The situation represented by *Voluntary* scenario causes conservation-willing owners' lands to be excluded from protection, which is not any fairer than enforcing conservation-resistant landowners to protect. In the Finnish context, especially owners of open mires may even lose an only opportunity to earn money with their mire property since the classification of mires' naturalness prevents e.g. peat mining in the pristine mires (Council of State 2012). In a wider context, the society loses conservation opportunities when lands of conservation-minded owners cannot be protected.

Obligatory scenario's cost-efficiency was high. Compared to two other scenarios, however, *Obligatory* scenario suffered from a high resistance by landowners. If implemented in practice, it would probably cause conservation conflicts and be therefore potentially detrimental to conservation.

Of the studied conservation scenarios, *Balancing* scenario seemed to least compromise the ecological gains, social acceptability, and economic costs all combined. Its cost-efficiency was intermediate compared to two other scenarios. It was able to achieve practically the same average feature representation as *Obligatory* scenario, while it allocated significantly less lands of conservation resistant landowners to protection.

4.3.2 Involuntary conservation implementation

Regardless of *Balancing* scenario's ability to best alleviate the trade-offs considering the studied conservation scenarios, it should be noted that it is worthy of its name mainly from a wider societal perspective. For a single landowner, protection is always a binary matter: it either happens or not.

Therefore, for resisting landowners, no matter how few they are, enforcing protection always represents an extreme solution.

Attitudes toward land expropriations for conservation purposes can be sharply negative. "*I am strongly against expropriation as an approach, especially in democratic societies. This philosophy returns the conservation community to the 'dark old days' of imperialist command-and-control approaches to achieving conservation goals.*", wrote an editor from the journal Conservation Letters when rejecting the manuscript of the chapter III without peer review. The attitude is somewhat understandable given the numerous command-and-control based conservation initiatives implemented especially in developing countries that are often organized by foreign actors and that have led to displacement of local residents from their homes and/or prevented their access to essential resources (Adams and Hutton 2007, Lele *et al.* 2010). Nevertheless, land expropriations are implemented for sake of many large initiatives that are considered as societally necessary ones in democratic decision-making processes so in that sense, involuntary conservation approaches are not more inherently wrong than voluntary conservation approaches are inherently right.

As Kamal *et al*. (2015) wrote in their review about conservation strategies, expropriations do not always mean displacement of locals or a full ban of resource use. In Finland, for instance, broad public rights of access (in Finnish 'jokamiehen oikeudet') ensure that in most cases, all citizens are allowed to enter the land and e.g. to pick berries and mushrooms, whether the land is owned by a private actor or the government, and whether the land is protected or not (Ministry of the Environment 2016). Also hunting can be allowed on protected sites if it does not jeopardize conservation values (Nature Conservation Act 1096/1996). In practice, basically, a landowner of a protected site loses her/his right to e.g. construct buildings or roads, log trees, ditch forests or mires, or otherwise alter the site on such ways that harm its conservation values. Additionally, the government is obligated to compensate economic losses caused by protection, whether the site is protected voluntarily or via expropriation. From landowners' point of view, expropriations for conservation purposes in Finland have been less restrictive than expropriations for many infrastructure initiatives as people have not been displaced or fully prohibited to utilize natural resources in consequence of conservation. Finnish compensation practices and public rights of access may alleviate the experienced harm caused by enforced protection and even reduce pre-emptive behavior (II) that has been shown to be a problem in USA and Australia (e.g. Lueck and Michael 2003, Simmons *et al.* 2018a).

4.3.3 Voluntary conservation implementation

Many studies show that different forms of voluntary conservation approaches facilitate acceptance of conservation amongst local people and make protection more successful in the long run (e.g. Brooks *et al.* 2006, Paloniemi and Tikka 2008). Hence, if sites owned by conservation-resistant landowners are desired to be added to a protected area network, then they should preferably be protected via voluntary means.

Conservation willingness can be boosted e.g. by improving landowners' knowledge and convincing them of high ecological value of their land, since they may not know or understand the lack of compensatory sites concerning endangered biodiversity features (Olive and Mccune 2017). Performance curves produced by Zonation (Figs. 5 and 10) can serve as a tool to make the trade-offs concrete to stakeholders by explicitly visualizing them (I).

Further increasing the level of financial compensation could persuade unwilling landowners to become more receptive to conservation (Sorice *et al.* 2013), though it may not satisfy all landowners, since other than economic reasons such as place attachment (Selinske *et al*. 2015) or dislike of government regulation (Olive and McCune 2017) cannot so easily be compensated with money. However, cost differences between *Voluntary* and two other scenarios raise an interesting option to try to increase resistant landowners' conservation willingness by paying more. If a market price is paid for each protected mire, the costs of *Obligatory* and *Balancing* scenarios are significantly lower than those of *Voluntary* scenario (Fig. 10). This means that both in *Obligatory* and *Balancing* scenarios, landowners of the most biodiversity-rich mires resisting protection could be paid even multifold compensations before the scenarios' total costs would match those of *Voluntary* scenario. Additionally, conservation-minded landowners could sell their land to protection for a relatively low price. This further opens up possibilities to save conservation costs in certain sites and allocate the savings to increase the likelihood of unwilling owners to participate in CMPP. One option could be opening the candidate mires proposed for CMPP for a conservation auction (Hanley *et al.* 2012). It would reveal landowners' opportunity costs and potentially enable protection of certain sites for lower than a market price, saving money compared to the current CMPP compensation practice. Due to the fragmented landownership, it is likely that the acceptable auction bids would cover mire entities unevenly, letting unprotected sites to exist in single mires. After the auction, there would be at least two options to complement the coverage of protection. First, in site-specific negotiations, the price is agreed between a conservation authority and a landowner. Second, an agglomeration bonus is a mechanism where a conservation authority designs a bonus, which is paid on top of the basic price, if the protected estate fulfills certain conditions such as touches another protected estate (Parkhurst *et al.* 2002). This creates an incentive to set aside adjacent estates, regardless of who owns them, therefore making cooperation between neighboring landowners beneficial. Theoretically, the mixture of at least two voluntary conservation mechanisms, the auction and negotiations and/or an agglomeration bonus could cost-efficiently resolve the challenges that the fragmented landownership and the large share of privately owned area cause in voluntary protection of mires or other such ecosystems that require spatial continuity.

Practically, however, in the case of CMPP there would be obstacles concerning the above-suggested mechanisms. The regulations of state subsidies set by the European Union prohibit national governments to pay for conservation more or less than income losses are, therefore preventing the government to implement competitive bidding (such as a conservation auction) or nature values trade (i.e. paying substantially more than a market price) (Hänninen *et al*. 2017). To enable more effective voluntary conservation practices, the European Union legislation should be revised. In the current situation in Finland, the government is by far the most important actor in procuring lands for protection, but it has few tools to protect spatially and structurally well-connected habitat ensembles by voluntary means. The role of private conservation foundations and crowdfunding is still small in Finland so the most important means to implement voluntary mire conservation will be landowner-specific negotiations and persuasion. In the prevailing circumstances, voluntary mire conservation seems challenging.

4.4 Critique of the research

The worst uncertainty of this thesis relates to the lack of reference cases and proper controls. It is impossible to know whether the CMPP decision-making process could have been even more successful than it was (I). The Zonation analysis does not describe e.g. factors affecting people behavior that likely had an influence on the success of the decision-making process, so alternative outcomes can just be speculated based on gut-feelings of the members in the working group (many of whom were authors in the chapter I). Similarly, pre-emptive behavior on CMPP candidate wooded mires was studied by comparing them to otherwise similar, but non-candidate wooded mires, which do not serve as a true control group (II). To make the study design to be more reliable, the control group should have been combined from such owners of the candidate wooded mires that did not get information about their mires belonging to the CMPP candidates, but such did not exist because all the owners were informed. Furthermore, reasons for the lack of pre-emptive behavior were not studied, so they can just be speculated. Both chapters I and II reflect the common challenges when studying real-world conservation cases: science is made on the framework that contemporary practical conservation cases happen to provide. Thus, utilization of the best scientific methods is often not possible. However, the CMPP decision-making process undeniably was successful because it provided an ecologically good outcome and was well accepted as a process and as an outcome (I). Similarly, even if some other factors than the lack of pre-emptive behavior affected harvesting rates on candidate wooded mires, the total harvesting rates per five study years and the monthly harvesting rates still showed that landowners did not execute systematic pre-emptive loggings (II).

In the research considering ecological, social, and economic trade-offs in CMPP, there was a need to extrapolate landowners' preferences, since using only the observed survey data would have restricted the analysis area impractically small for prioritization analyses (III). Accordingly, the results of the chapter III cannot be applied directly to the conservation planning in a sense that the combinations of mires excluded from and included in protection in each scenario are not real combinations, but estimations. However, the results very likely reflect the real life conservation situations, since sample sizes were large and extrapolations randomized according to certain known facts such that landowners' attitudes towards conservation can depend on their relationship to regional environmental authorities (Salomaa *et al*. 2016).

Nevertheless, one should be cautious in generalizing the results of this thesis. Especially the lack of pre-emptive behavior on CMPP wooded mires can be a case-specific issue, because results drawn from other countries prove that pre-emptive behavior is a true phenomenon (Brook *et al.* 2003, Lueck and Michael 2003, Zhang 2004, Simmons *et al.* 2018b) (II). It is even possible that the results concerning pre-emptive behavior could have been different if the study was implemented on mineral soil forests in Finland, because silvicultural value of peatland forests is on average lower than that of mineral soil forests (II), which could affect landowners' behavior. Results of ecological, social, and economic trade-offs in CMPP and possibilities to alleviate them can be generalized for protecting such ecosystems or species, which require large continuous areas to maintain their features or populations (III). However, it is uncertain whether the trade-offs would have been as evident if they were studied in ecosystems where spatial continuity does not have equally significant role.

Overall, the ecological and natural-scientific aspects seen in the aims and methods of this thesis are not exclusive enough to cover diversity of the themes shown by the chapters I–III. In order to deepen the results and conclusions drawn from the thesis, methods e.g. from social, political, or psychological sciences should have been utilized.

5 CONCLUSIONS

Success or failure of boreal mire conservation in Finland may seem trivial, but this thesis shows that it reflects the global state of nature and challenges of conservation. Boreal mire conservation is intertwined with diverse stakeholder groups (I), hard and unavoidable social-ecological trade-offs (III), and multilevel interrelationships between ecological, societal, and economic matters (II, III). It is as much a political and a societal matter as a natural-scientific exercise (I, II, III).

The first two questions of this thesis asked how to build a spatial prioritization analysis to take into account multiple ecological and societal needs, and how to make spatial prioritization and conservation decision-making to support each other and fill the science-practice gap. Although the first question seems purely a technical issue, while the second one is about decision-making practices, they cannot be answered without simultaneously considering the both.

An ecologically successful CMPP conservation plan was produced in cooperation between prioritization experts and diverse stakeholders, filling the science-practice gap (I). Although the spatial prioritization method had a significant role in the plan's ecological success, it is likely that the result would not have been that good without participation of the stakeholders. By defining the aims of the prioritization analysis and sharing their knowledge, the stakeholders pushed the prioritization experts to take most out of the technical characteristics of the prioritization method. Simultaneously, the method enabled sharing and analytical use of knowledge between the prioritization experts and the stakeholders, which is far more than could be expected from a purely technical prioritization tool. I conclude that in the CMPP planning process, the science-practice gap was filled because of two main factors. First, stakeholders and experts cooperated. Second, the cooperation was centered on the scientifically valid, high-end systematic prioritization method.

The latter two questions of this theses asked whether the harvesting rates of wooded CMPP candidate mires differed from those of non-candidate ones, and how acknowledging or ignoring landowners' conservation preferences in conservation planning can influence ecological gains and economic costs of conservation. Answers to these contribute also to the construction of a successful prioritization analysis and utilizing it in conservation decision-making.

The original political decision to allow land expropriations in CMPP without considering landowners' conservation preferences could provoke negative attitudes toward conservation (III). However, the fright concerning intentional destruction of biodiversity values aiming to avoid protection seems to be unfounded in this case (II). The lack of systematic pre-emptive loggings can be a result of the societal characteristics more or less specific for Finland. Comprehensive compensation practices together with broad public rights of access can at least partly explain the encouraging result. At the time of making the decision about allowing expropriations, it was known that they would simplify protection of large, continuous, and mostly privately-owned areas, but the decision's effect on pre-emptive behavior was not known. Hence, the decision's impacts on biodiversity could have been more harmful than what they finally were.

The latter political decision to reject land expropriations in CMPP without providing more tools for voluntary conservation causes significant ecological and economic costs (III). Moreover, Helmi-program established to protect mires voluntarily has a target to set aside 20 000 ha by 2023 (https://www.ym.fi/helmi). If the target was reached, totally 60 000 ha or about 51% of the original 117 000 ha target would be set aside by 2023, so at least in short term, ecological costs of the rejection can be larger than shown in chapter III. However, the ecological costs should be carefully analyzed because especially open mires can be hard to transfer into anthropogenic use, meaning that many of CMPP candidate mires may not be destroyed even without protection. On the other hand, mires' hydrology easily suffers also due to land use actions that are not made precisely on those mires, but on their near surroundings (Tahvanainen 2011), so exclusion from protection may expose mires to drying. Furthermore, CMPP candidate wooded mires are continuously harvested, albeit slower than other wooded mires (II). Whether one is for or against land expropriations for conservation or any other purposes, it can be concluded that rejecting the option of expropriations was an epitome of a precipitate decision, the consequences of which were not known in the moment of making the decision, and the consequences of which happened to be larger than was probably expected (III).

Regardless of whether conservation is implemented voluntarily or involuntarily, alleviation of the trade-offs between ecological, social, and economic matters would be possible, but it would require changes to the current practices (III). First, the alternative solutions should be detected and their consequences studied. When originally establishing CMPP, the ecological considerations were set as the main aim (I) as they usually are in nature conservation. Decision-makers also allowed land expropriations (Council of State 2012), but obviously, they did not recognize alternatives for this. Therefore, the prioritization analysis striving to reconcile ecological considerations and landowners' conservation preferences was not even thought about, although it would have significantly reduced the need for expropriations without compromising the ecological aims (III). Correspondingly, deciding about voluntary protection before starting the planning process likely would have prevented even thinking about protecting conservation-resistant owners' lands although it would have opened up ideas for more effective resource allocation and enabled larger ecological gains compared to categorically excluding conservation-resistant landowners' lands. Detecting the alternatives would have allowed acknowledging the resistance but not letting it overrule all the other aspects, leading to ecologically, socially, and economically better outcome regardless of chosen conservation approach.

The second prerequisite for alleviation of the trade-offs is that the current European Union legislation should be revised to allow paying for biodiversity values more or less than landowners' income losses are. This would be essential for the governments of the European Union countries to be able to implement voluntary conservation efficiently. Another option would be strengthen the role of private conservation agents, but in Finland the government has traditionally had a significant role in nature conservation and compensations have been paid mostly from tax revenue, so it is improbable that private agents could take a significantly larger role in conservation at least in a short time scale.

I conclude that none of the decisions, neither e.g. certain areal targets, nor conservation means should be fixed before consequences of alternatives are studied (see also e.g. Wilhere 2008). Pre-fixing easily prevents to detect alternative solutions and therefore, finding the most suitable one. I further conclude that spatial prioritization methods provide a decent tool to visualize interrelationships between different ecological, social, and economic factors and their relative changes (III). Therefore, they can be used to inform stakeholders and decision-makers about the consequences of alternative conservation solutions, potentially increasing the likelihood of science-practice gap to be fulfilled (I, III).

Ultimately, I argue that the foremost reasons for the challenges faced by nature conservation are system-related. Interactions between distant ecological and socioeconomic systems, flows, agents, causes, and effects are nowadays a global norm (Liu *et al.* 2013). As an example, production and consumption of resources for people are often separated far from each other both geographically and temporally (Díaz *et al.* 2019, Gardner *et al.* 2019). These distant interactions detach us from nature and prevent us to see the consequences of our actions and the importance of conservation. It is possible that the anthropocentric view that permeates also the concept of Nature's Contributions to People (Díaz *et al.* 2018) still does not comprehensively enough describe the complex dependence of human societies on the other life. I argue that we should consider and appreciate also nature's contributions to other species as passionately as to our own human species, and start to act accordingly. We can squelch on the boreal mire, have a coffee in Starbucks, or sit at a laptop writing science, but still we breathe the air ecosystems have produced. The clothes we wear, the coffee we swallow, and the keyboard we touch are products of nature. Nature is everywhere. Only becoming deeply conscious of this we can understand the value of nature and its conservation.

Although people have make this journey worth of going, the sponsors have enabled the journey. I thank Maj and Tor Nessling Foundation for granting the PhD work of the young Master of Science, who was scared of her future being the one of a long-term unemployed biologist. I also thank Kone Foundation for granting the last year of my PhD work.

Then the people, and very first my three supervisors. Janne, I always felt that you believed me. Even during those times when I was often about to quit, I felt you were with me. You have also shown that a scientist can and should take part to the societal dialogue. In that sense, your courage and example have inspired me a lot, and your thoughts about the soul of debating have greatly directed my thinking. You have shown that debating is not about winning or being right. It is about getting closer to the truth. "*Riittävän hyvillä perusteilla olen aina valmis kääntämään takkini.*"

Panu, you have held up the everyday life of a PhD student. You always answer when I call. In many weekends and nights, I have sent e-mails with caps lock titles to all of you and very often, you have been the one who first answers. You have fought for the rights of grant researches like fury and supported me when the institutions have treated me unfairly. You have shown that if something is wrong, things simply need to be fixed. And you have inspired me to act on that way, too.

Santtu, I have felt that at least every now and then, you have been thrilled about supervising me! When starting my PhD work, one aim was that I would learn to use Zonation. You have really done your best in teaching me. But even more importantly, you have given a lot of positive feedback to me, which have carried me many weeks, even months at a time. Here, your example affected how I supervised my own master student.

I guess that considering your supervision, Janne has had the easiest times, Panu the hardest, and Santtu the funniest ones!

I thank the members of my follow-up group, Senior Ministerial Adviser Aulikki Alanen, Emeritus Professor Yrjö Haila, and FT Risto Sulkava. Discussions with you cast faith that my thesis will be good and in time. I also thank the reviewers of the thesis Doctor Jonas Geldmann and Research Professor Matti Koivula, who gave very constructive criticism, therefore greatly helping me in preparing to the public defense.

It is a pleasure to have Doctor Anni Arponen as my opponent! You have authored many papers that I cite in my thesis. I appreciate especially your skills and research in spatial conservation prioritization.

Many institutions and their people have had a role in my PhD work. I thank the Ministry of the Environment: Aulikki Alanen afforded research data to me, and Aulikki together with Kaisu Aapala patiently answered to my questions. Thanks also to Juha Väisänen in Finnish Forest Centre for providing forest-owner data. Many thanks to Metsähallitus for allowing Santtu Kareksela to supervise me, and for people working in Metsähallitus, who assisted me in many kinds of problems: Marja Hokkanen, Sakari Rehell, Suvi Haapalehto, and others. I also thank people in ELY Centres, who helped me with the landowner survey data: Kristiina Nyholm, Johanna Hallman, and others. And thank you Anna Salomaa in University of Helsinki and Riikka Paloniemi in Finnish Environment Institute for thought-provoking discussions concerning landowners' motivations.

I also thank Kalle Salovaara, whom Master's thesis I had a pleasure to supervise. I am grateful to your huge work in combining the data and finding out the best way to make the statistical analyses in the chapter II.

AaKoo, Anna, Illi, Jani, Jenna, Jenni, Kaisa R., Kaisa T., Kristiina, Linda, Maiju, Merja, and Tähti. I have never had such a work community as you, and I never will. You are so much more than just colleagues. You are my friends. We have broken many taboos in academic working culture and I hope that in this regard, we will spread good around us, no matter where we will work in the future.

I am sure that some Moomins from my valley would have escaped without Pätkä. Nothing recovers exhausted human mind like doing things with animals. Here, Salla deserves special thanks for bringing so much content to my dog activities! Or, should I even say, to *our* dog activities.

Ankka, Johanna, Mari, Sini, Såkke. My old friends, kalsarikännääminen with you is worth of buying alcohol! And COVID-19 restrictions has proved that kalsarikännääminen with you goes fine also via Skype. You may have not noticed, but you affected my career already years ago. With you, I first discussed about the society, politics, economy, environment, religion, and other forces that move the world and lives of all living creatures. I really appreciate your thoughts and a possibility to discuss in a safety athmosphere.

Sami. You know me better than anyone else and still, you renovate my apartment, sometimes take Pätkä to an evening walk so that I can go to the gym, listen to my anxiety about life, and last but not least, cook for me! There are no words to describe it properly, so unfortunately, only I know how significant this all has been to me, and still is.

Ekku. Thank you for being the most reliable person in my life. With you, I never need to pretend anything. Our long walks on forest tracks and discussions in sauna are very precious moments for me. If only we could have them more often!

Äiti ja isi. Kiitos, että voin koska tahansa tulla kotiin, jossa asiat ovat niin kuin ennenkin. Ruoka pöydässä ja sauna lämpimänä. Huolimatta saavuttamastani tohtorin tittelistä, olen lopulta hyvin yksinkertainen ihminen ja onnellisimmillani silloin, kun saan riittävästi ruokaa, lepoa ja liikuntaa.

YHTEENVETO (RÉSUMÉ IN FINNISH)

Miten suojella luontoa – Tapauksena soiden suojelu Suomessa

Ihmiskunnan olemassaolo ja tulevaisuus ovat täysin riippuvaisia luonnon ekosysteemien toiminnasta. Olemme kuitenkin valjastaneet luonnon niin tehokkaasti omaan käyttöömme, että ekosysteemien toiminnot ovat alkaneet heiketä ja muuttua. Tämä on johtanut monien lajien uhanalaistumiseen ja katoamiseen sekä omien yhteiskuntiemme tulevaisuuden vaarantumiseen.

Luonnonsuojelualueet hidastavat ja ehkäisevät ekosysteemien heikentymistä ja lajien uhanalaistumista. Kansainväliset sopimukset osoittavat, että luonnon monimuotoisuuden turvaamiseen on ainakin näennäisesti olemassa poliittista tahtoa. Silti luonnonsuojelu on usein vaikeaa. Monesti suojelun hyödyt ja haitat jakaantuvat epätasaisesti eri ihmisryhmien välille sekä ovat keskenään yhteismitattomia. Tästä seuraa helposti konflikteja ja eturistiriitoja.

Tässä väitöskirjassa tutkin, miten luonnonsuojelun päätöksenteko voisi edistää ekologisesti, yhteiskunnallisesti ja taloudellisesti kestävää suojelua. Tutkimustapauksena käytin soidensuojelun täydennysohjelmaa. Vuonna 2012 tehtyyn poliittiseen päätökseen perustuen ohjelma oli tarkoitus toteuttaa luonnonsuojeluohjelmana, jossa maiden lunastaminen suojelutarkoituksiin oli sallittua. Taustalla vaikutti se tosiasia, että suoluonto on riippuvainen koko suoaltaan veden pinnan tasosta: mikäli osa suosta esimerkiksi ojitetaan tai hakataan, kuivuminen uhkaa koko suota. Tämän vuoksi suot on suojeltava vesitaloudellisina kokonaisuuksina, jotka ovat pinta-alaltaan usein varsin suuria. Koska maanomistajuus on Suomessa hyvin sirpaloitunutta, voi yhdellä suolla olla jopa yli sata omistajaa. Niinpä maan lunastusten salliminen olisi ollut helppo tapa suojella isojakin suokokonaisuuksia. Vuonna 2014 poliittinen ilmapiiri kuitenkin muuttui, eikä maan lunastuksia ohjelman piirissä sittenkään sallittu. Koska hallinnollisia työkaluja soiden vapaaehtoiseen suojeluun ei ollut, ja koska luonnonsuojelun resursseja sittemmin leikattiin tuntuvasti, pysähtyi ohjelman toteuttaminen lukuun ottamatta valtion omistamia soita. Tällä hetkellä ohjelman toteutus jatkuu Helmi-elinympäristöohjelman puitteissa maanomistajien vapaaehtoisuuteen perustuen, joskin Helmen lyhyen aikavälin tavoitteet ovat alkuperäistä soidensuojelun täydennysohjelmaa maltillisemmat.

Täydennysohjelman suojelusuunnitelma ja suovalinnat tehtiin siis kysymättä maanomistajien suojeluhalukkuutta tai -haluttomuutta, koska maan lunastusten piti mahdollistaa minkä tahansa suon suojelu. Ohjelman suunnittelu onnistui hyvin. Ohjelmaa valmistellut työryhmä päätti, että osa suojeltavista soista valittaisiin Zonationilla, joka on luonnonsuojelun priorisaation ja päätöksenteon ohjelma, ja loput suot pisteytysmenetelmää käyttäen. Zonation-analyysi tehtiin tiiviissä yhteistyössä työryhmän ja priorisaation asiantuntijoiden kesken. Analyysi ja soiden valintaa koskeva päätöksenteko tukivat toinen toistaan. Yhtäältä analyysin tekeminen ohjasi työryhmän jäsenet määrittelemään tarkoin soiden suojelun ekologiset tavoitteet sekä jakamaan ja käyttämään analyyttisesti ryhmän jäsenten suotietoa. Toisaalta työryhmän jäsenten määrittelemät tavoitteet ja heidän jakamansa tieto ohjasivat priorisaation asiantuntijoita hyödyntämään parhaalla mahdollisella tavalla Zonationin teknisiä ominaisuuksia. Lopputuloksena syntyi soidensuojelusuunnitelma, joka paikkasi tehokkaasti olemassa olevan suojelualueverkoston puutteita ja oli siten suoluonnon kannalta erittäin onnistunut. Usein priorisointimenetelmien rooli luonnonsuojelussa nähdään kapeasti siten, että ne ovat pelkkiä kohdevalinnan teknisiä työkaluja. Soidensuojelun täydennysohjelman suunnittelu kuitenkin osoittaa, että ne voivat parantaa suojelupäätöksenteon laatua monin eri tavoin. Parhaimmillaan ne mahdollistavat niin sanotun hiljaisen asiantuntijatiedon analyyttisen käytön, auttavat tiedon jakamisessa asiantuntijoiden kesken ja lopulta kaventavat kuilua tieteellisten menetelmien ja käytännön suojelutyön välillä.

Maan lunastusten salliminen alkuperäisessä täydennysohjelmassa aiheutti epäilyjä, että puustoisia soita saatettaisiin aavistushakata. Aavistushakkuulla tarkoitetaan hakkuuta, jonka maanomistaja tekee tuhotakseen tai heikentääkseen metsänsä luontoarvoja ja siten välttyäkseen maansa suojelulta. Tutkimukset Yhdysvalloissa ja Australiassa ovat osoittaneet, että erilaiset aavistavat toimet luonnon heikentämiseksi ovat todellinen ilmiö. Väitöskirjani tulokset kuitenkin osoittavat, että Suomessa täydennysohjelmaan ehdolla olleita puustoisia soita ei aavistushakattu ainakaan systemaattisesti. Syinä voivat olla esimerkiksi kattavat suojelun korvauskäytänteet, soiden keskimäärin vähäisempi metsätaloudellinen arvo verrattuna kivennäismaiden metsiin tai suomalaisten maanomistajien halu suojella soitaan.

Soidensuojelun täydennysohjelman muuttaminen vapaaehtoisuuteen perustuvaksi muutti radikaalisti myös suojelusuunnittelua. Jo tehty suojelusuunnitelma menetti merkitystään, koska maanomistajien sallittiin kieltäytyä suojelusta, eivätkä kaikki suot siten olleetkaan valittavissa suojeluun. Väitöskirjassani sisällytin maanomistajien suojeluhaluttomuuden ja suojelun taloudelliset kustannukset jo tehtyyn Zonation-analyysiin ja vertailin, millaisia ekologisia ja taloudellisia vaikutuksia maanomistajien mielipiteiden huomioiminen tai toisaalta huomiotta jättäminen aiheuttaa soiden suojelussa. Tulokset osoittavat, että suojelu on hyvin kustannustehotonta, mikäli suojelun ulkopuolelle jätetään järjestelmällisesti kaikki sellaiset suot, joilla yksikin maanomistaja vastustaa suojelua. Tämä maksaa paljon rahaa, vaikka suojellun monimuotoisuuden määrä jää vaatimattomaksi. Maanomistajien mielipiteiden huomiotta jättäminen taas mahdollistaa kustannustehokkaan suojelun, mutta tällöin suojeluun päätyy runsaasti myös suojelua vastustavien maanomistajien soita. Parhaiten eri tarpeita pystytäänkin huomioimaan siten, että maanomistajien mielipiteiden annetaan vaikuttaa soiden valintaan, mutta monimuotoisuudeltaan erityisen rikkaat kohteet suojellaan, vaikka niiden omistajat vastustaisivatkin suojelua. Tällöin taloudelliset kustannukset pysyvät kurissa, vaikka suojellun monimuotoisuuden määrä säilyy korkeana. Lisäksi valtaosin vältytään suojelemasta haluttomien maanomistajien soita.

Tämä ei automaattisesti tarkoita sitä, että maan lunastukset olisivat välttämättömiä. Sen sijaan tutkimustulokset avaavat kiintoisia mahdollisuuksia suojeluresurssien tehokkaampaan kohdentamiseen. Yksi vaihtoehto olisi pyrkiä suojelemaan monimuotoisuudeltaan rikkaat, mutta suojelua vastustavien tahojen omistamat suot vapaaehtoisin keinoin yksinkertaisesti korottamalla suojelukorvausta. Vastustetuista soista voitaisiin maksaa tuntuvasti nykyistä enemmän, ennen kuin kokonaiskustannukset kohoaisivat yhtä suuriksi kuin siinä tapauksessa, että kaikki vastustetut suot jätettäisiin automaattisesti suojelun ulkopuolelle. Tällaisesta niin sanotusta luonnonarvokaupasta saatiin hyviä kokemuksia 2000-luvun alkupuolella, jolloin sitä kokeiltiin Etelä-Suomen metsien monimuotoisuusohjelma Metsossa. Sittemmin Euroopan Unionin valtiontukisäännöt sallivat maksuperusteeksi ainoastaan tulonmenetykset, estäen luonnonarvokaupan. Voimassa oleva lainsäädäntö siis omalta osaltaan estää kustannustehokkaan ja hyväksyttävän luonnonsuojelun toteuttamista.

Sekä vapaaehtoinen että maan lunastuksiin perustuva luonnonsuojelu ovat molemmat arvovalintoja. Siksi poliitikkojen tehtävä on päättää, millä keinoin luontoa lopulta suojellaan. Ekologisten, taloudellisten ja sosiaalisten näkökulmien välisiä ristiriitoja on kuitenkin mahdollista lieventää niin vapaaehtoisessa kuin maan lunastukset sallivassa suojelussa. Tämä kuitenkin edellyttää, että vaihtoehtoiset suojeluratkaisut huomataan ja niiden vaikutukset analysoidaan jo ennalta. Kun maan lunastukset alun perin päätettiin sallia soidensuojelun täydennysohjelmassa, oli vaarana, että puustoisten soiden monimuotoisuutta tuhoutuisi mahdollisesti laajamittaisiksi yltyvissä aavistushakkuissa. Tämän uhan väitöskirjani osoittaa aiheettomaksi, mutta yhtä kaikki päätös sallia maan lunastukset tehtiin sokkona, koska tietoa aiheesta ei vielä tuolloin ollut saatavilla. Lisäksi maan lunastusten salliminen johti siihen, että kun soita valittiin täydennysohjelmaan, ei maanomistajien mielipiteitä kysytty. On mahdollista, ettei toista vaihtoehtoa eli mielipiteiden kysymistä edes huomattu. Näin ollen ei myöskään huomattu, että maanomistajien mielipiteiden kysyminen olisi johtanut vähälukuisempiin lunastuksiin, suojellun monimuotoisuuden määrästä kuitenkaan tinkimättä. On myös mahdollista, että jos suot olisi alun perinkin päätetty suojella vapaaehtoisesti, olisi suojelua vastustaneiden maanomistajien maat jätetty järjestelmällisesti suojelun ulkopuolelle, mikä taas olisi johtanut kustannustehottomaan soiden suojeluun. Olisikin tärkeää, etteivät poliittiset arvovalinnat vaikuttaisi suojeltavien kohteiden tekniseen valintaan. Tällöin voitaisiin aidosti analysoida eri suojeluvaihtoehtojen vaikutuksia. Poliitikot saisivat tietoa päätöksiensä tueksi ja poliittisten päätösten vaikutukset tulisivat näkyviksi myös suurelle yleisölle. Tämä mahdollistaisi ekologisesti, taloudellisesti ja yhteiskunnallisesti hyväksyttävämmän luonnonsuojelun.

REFERENCES

Adams V.M., Pressey R.L. & Naidoo R. 2010. Opportunity costs: Who really pays for conservation? *Biol. Conserv.* 143: 439–448.

Adams V.M., Pressey R.L. & Stoeckl N. 2014. Estimating landholders' probability of participating in a stewardship program, and the implications for spatial conservation priorities. *PLoS ONE* 9. doi:10.1371/journal.pone.0097941

Adams W.M. & Hutton J. 2007. People, Parks and Poverty: Political Ecology and Biodiversity Conservation. *Conserv. Soc.* 5: 147–183, doi:https://www.jstor.org/stable/2639287.

Alanen A. & Aapala K. (eds.). 2015. *Proposal of the Mire Conservation Group for supplemental mire conservation*. Reports of the Ministry of the Environment 26 | 2015. Available from: http://hdl.handle.net/10138/158285

Alphandéry P. & Fortier A. 2001. Can a territorial policy be based on science alone? The system for creating the Natura 2000 network in France. *Sociol. Ruralis* 41: 311–328.

Bain C.G., Bonn A., Stoneman R., Chapman S., Coupar A., Evans M., Gearey B., Howat M., Joosten H., Keenleyside C., Labadz J., Lindsay R., Littlewood N., Lunt P., Miller C.J., Moxey A., Orr H., Reed M. Smith P., Swales V., Thompson D.B.A., Thompson P.S., Van de Noort R., Wilson J.D. & Worrall F. 2011. IUCN UK Commission of Inquiry on Peatlands. IUCN UK Peatland Programme, Edinburgh. Available from: https://urly.fi/1rgt

Ball I.R., Possingham H.P. & Watts M.E. 2009. Marxan and relatives: Software for spatial conservation prioritization. In: Moilanen A., Wilson K.A. & Possingham H.P. (eds.), *Spatial Conservation Prioritization - Quantitative Methods and Computational Tools*, Oxford University Press, New York, pp. 185–195.

Balmford A., Bruner A., Cooper P., Costanza R., Farber S., Green R.E., Jenkins M., Jefferiss P., Jessamy V., Madden J., Munro K., Myers N., Naeem S., Paavola J., Rayment M., Rosendo S., Roughgarden J., Trumper K. & Turner R.K. 2002. Economic reasons for conserving wild nature. *Science* 297: 950–953.

Balmford A. & Whitten T. 2003. Who should pay for tropical conservation, and how could the costs be met? *Oryx* 37: 238–250.

Bar-On Y.M., Phillips R. & Milo R. 2018. The biomass distribution on Earth. *PNAS* 115: 6506–6511.

Batavia C., Nelson M.P. & Wallach A.D. 2020. The moral residue of conservation. *Conserv. Biol*. doi:https://doi.org/10.1111/cobi.13463

Bennett N.J. & Dearden P. 2014. Why local people do not support conservation: Community perceptions of marine protected area livelihood impacts, governance and management in Thailand. *Mar. Policy* 44: 107–116.

Bertuol-Garcia D., Morsello C., N. El-Hani C. & Pardini R. 2018. A conceptual framework for understanding the perspectives on the causes of the science–practice gap in ecology and conservation. *Biol. Rev.* 93: 1032–1055.

Beyer H.L., Dujardin Y., Watts M.E. & Possingham H.P. 2016. Solving conservation planning problems with integer linear programming. *Ecol. Model.* 328: 14–22.

Blicharska M., Orlikowska E.H., Roberge J.M. & Grodzinska-Jurczak M. 2016. Contribution of social science to large scale biodiversity conservation: A review of research about the Natura 2000 network. *Biol. Conserv.* 199: 110–122.

Blicharska M., Smithers R.J., Mikusiński G., Rönnbäck P., Harrison P.A., Nilsson M. & Sutherland W.J. 2019. Biodiversity's contributions to sustainable development. *Nat. Sustain.* 2: 1083–1093.

Blondet M., Koning J. de, Borrass L., Ferranti F., Geitzenauer M., Weiss G., Turnhout E. & Winkel G. 2017. Participation in the implementation of Natura 2000: A comparative study of six EU member states. *Land Use Policy* 66: 346–355.

Brander B.J.A. & Taylor M.S. 1998. The Simple Economics of Easter Island: A Ricardo-Malthus Model of Renewable Resource Use. *Am. Econ. Rev.* 88: 119–138.

Brondo K.V. & Bown N. 2011. Neoliberal conservation, Garifuna territorial rights and resource management in the Cayos Cochinos Marine Protected Area. *Conserv. Soc.* 9: 91–105.

Brook A., Zint M. & Young R. De. 2003. Landowners' Responses to an Endangered Species Act Listing and Implications for Encouraging Conservation. *Conserv. Biol.* 17: 1638–1649.

Brooks J.S., Franzen M.A., Holmes C.M., Grote M.N. & Mulder M.B. 2006. Testing hypotheses for the success of different conservation strategies. *Conserv. Biol.* 20: 1528–1538.

Byl J.P. 2019. Perverse Incentives and Safe Harbors in the Endangered Species Act: Evidence From Timber Harvests Near Woodpeckers. *Ecol. Econ.* 157: 100–108.

Cardinale B.J., Duffy J.E., Gonzalez A., Hooper D.U., Perrings C., Venail P., Narwani A., MacE G.M., Tilman D., Wardle D.A., Kinzig A.P., Daily G.C., Loreau M., Grace J.B., Larigauderie A., Srivastava D.S. & Naeem S. 2012. Biodiversity loss and its impact on humanity. *Nature* 486: 59–67.

CBD. 1992. Convention on Biological Diversity. United Nations.

CBD. 2011. Aichi Target 11. Decision X/2. Convention on Biological Diversity.

Chan K.M.A., Guerry A.D., Balvanera P., Klain S., Satterfield T., Basurto X., Bostrom A., Chuenpagdee R., Gould R., Halpern B.S., Hannahs N., Levine J., Norton B., Ruckelshaus M., Russell R., Tam J. & Woodside U. 2012. Where are Cultural and Social in Ecosystem Services? A Framework for Constructive Engagement. *BioScience* 62: 744–756.

Colchester M. 2004. Conservation policy and indigenous peoples. *Environ. Sci. Pol.* 7: 145–153.

Costanza R. & Daly H.E. 1992. Natural capital and sustainable development. *Conserv. Biol.* 6: 37–46.

Costanza R., D'Arge R., Groot R. de, Farber S., Grasso M., Hannon B., Limburg K., Naeem S., O'Neill R.V., Paruelo J., Raskin R.G., Sutton P. & Belt M. van den. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387: 253–260.

Council Directive 92/43/EEC. On the conservation of natural habitats and of wild fauna and flora. Available from: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:31992L0043

Council of State. 2012. *The Government Resolution on the Sustainable Use and Protection of Peatlands*. Available in Finnish from: https://urly.fi/1mu5

Council of State. 2019. Luonnonsuojeluun merkittävä lisärahoitus budjettiriihestä. Announcement 17.9.2019. Ministry of the Environment. Available from: https://urly.fi/1sK1 (accessed in February 2020).

Cronon W. 1995. The trouble with wilderness; or, getting back to the wrong nature. In: Cronon W. (eds.), *Uncommon Ground: Toward Reinventing Nature*, New York: W.W. Norton & Co, pp. 69–90.

Crooks K.R. & Sanjayan M. 2006. *Connectivity conservation*. Cambridge University Press, Cambridge.

Díaz S., Demissew S., Carabias J., Joly C., Lonsdale M., Ash N., Larigauderie A., Adhikari J.R., Arico S., Báldi A., Bartuska A., Baste I.A., Bilgin A., Brondizio E., Chan K.M.A., Figueroa V.E., Duraiappah A., Fischer M., Hill R., Koetz T., Leadley P., Lyver P., Mace G.M., Martin-Lopez B., Okumura M., Pacheco D., Pascual U., Pérez E.S., Reyers B., Roth E., Saito O., Scholes R.J., Sharma N., Tallis H., Thaman R., Watson R., Yahara T., Hamid Z.A., Akosim C., Al-Hafedh Y., Allahverdiyev R., Amankwah E., Asah T.S., Asfaw Z., Bartus G., Brooks A.L., Caillaux J., Dalle G., Darnaedi D., Driver A., Erpul G., Escobar-Eyzaguirre P., Failler P., Fouda A.M.M., Fu B., Gundimeda H., Hashimoto S., Homer F., Lavorel S., Lichtenstein G., Mala W.A., Mandivenyi W., Matczak P., Mbizvo C., Mehrdadi M., Metzger J.P., Mikissa J.B., Moller H., Mooney H.A., Mumby P., Nagendra H., Nesshover C., Oteng-Yeboah A.A., Pataki G., Roué M., Rubis J., Schultz M., Smith P., Sumaila R., Takeuchi K., Thomas S., Verma M., Yeo-Chang Y. & Zlatanova D. 2015. The IPBES Conceptual Framework - connecting nature and people. *Curr. Opin. Env. Sust.* 14: 1–16.

Díaz S., Pascual U., Stenseke M., Martín-López B., Watson R.T., Molnár Z., Hill R., Chan K.M.A., Baste I.A., Brauman K.A., Polasky S., Church A., Lonsdale M., Larigauderie A., Leadley P.W., Oudenhoven A.P.E. van, Plaat F. van der, Schröter M., Lavorel S., Aumeeruddy-Thomas Y., Bukvareva E., Davies K., Demissew S., Erpul G., Failler P., Guerra C.A., Hewitt C.L., Keune H., Lindley S. & Shirayama Y. 2018. Assessing nature's contributions to people. *Science* 359: 270–272.

Díaz S., Settele J., Brondízio E.S., Ngo H.T., Agard J., Arneth A., Balvanera P., Brauman K.A., Butchart S.H.M., Chan K.M.A., Lucas A.G., Ichii K., Liu J., Subramanian S.M., Midgley G.F., Miloslavich P., Molnár Z., Obura D., Pfaff A., Polasky S., Purvis A., Razzaque J., Reyers B., Chowdhury R.R., Shin Y.J., Visseren-Hamakers I., Willis K.J. & Zayas C.N. 2019. Pervasive humandriven decline of life on Earth points to the need for transformative change. *Science* 366: eaax3100. doi:10.1126/science.aax3100

Donahue D. 2005. The Endangered Species Act and its current set of incentive tools for species protection. In Shogren J.F. (eds.), *Species at Risk: Using Economic Incentives to Shelter Endangered Species on Private Lands*, University of Texas Press, Austin, pp. 25–63.

Drescher M., Perera A.H., Johnson C.J., Buse L.J., Drew C.A. & Burgman M.A. 2013. Toward rigorous use of expert knowledge in ecological research. *Ecosphere* 4: 1–26.

Dudley N. 2008. (eds.) *Guidelines for Applying Protected Area Management Categories*. IUCN, Gland.

Endangered Species Act of 1973. U.S. Fish & Wildlife Service. Available from: https://www.fws.gov/international/pdf/esa.pdf (accessed February 2020).

Eurola S., Huttunen A., Kaakinen E., Kukko-oja K., Saari V. & Salonen V. 2015. Sata suotyyppiä. Opas Suomen suokasvillisuuden tuntemiseen. Thule-instituutti, Oulu.

Ferraro P.J., Mcintosh C. & Ospina M. 2007. The effectiveness of the US endangered species act: An econometric analysis using matching methods. *J. Environ. Econ. Manag.* 54: 245–261.

Ferrier S. & Wintle B.A. 2009. Quantitative approaches to spatial conservation prioritization: Matching the solutions to the need. In: Moilanen A., Wilson K.A. & Possingham H.P. (eds.), *Spatial Conservation Prioritization - Quantitative Methods and Computational Tools*, Oxford University Press, New York, pp. 1–15.

Finnish Forest Centre. 2018. Law enforcement of the Forest Acts 2018. Available in Finnish from:
https://www.metsakeskus.fi/sites/default/files/lainvalvonta-2018.pdf (accessed January 2020).

Gamfeldt L., Snäll T., Bagchi R., Jonsson M., Gustafsson L., Kjellander P., Ruiz-Jaen M.C., Fröberg M., Stendahl J., Philipson C.D., Mikusiński G., Andersson E., Westerlund B., Andrén H., Moberg F., Moen J. & Bengtsson J. 2013. Higher levels of multiple ecosystem services are found in forests with more tree species. *Nat. Commun.* 4: 1340.
https://doi.org/10.1038/ncomms2328

Gannon P., Dubois G., Dudley N., Ervin J., Ferrier S., Gidda S., Mackinnon K., Richardson K., Schmidt M., Seyoum-edjigu E. & Shestakov A. 2019. Editorial essay: An update on progress towards aichi biodiversity target 11. *Parks* 25: 7–18.

Gardner T. A., Benzie M., Börner J., Dawkins E., Fick S., Garrett R., Godar J., Grimard A., Lake S., Larsen R.K., Mardas N., McDermott C.L., Meyfroidt P., Osbeck M., Persson M., Sembres T., Suavet C., Strassburg B., Trevisan A., West C. & Wolvekamp P. 2019. Transparency and sustainability in global commodity supply chains. *World Dev.* 121: 163-177.

Geldmann J., Barnes M., Coad L., Craigie I.D., Hockings M. & Burgess N.D. 2013. Effectiveness of terrestrial protected areas in reducing habitat loss and population declines. *Biol. Conserv.* 161: 230–238.

Geldmann J., Coad L., Barnes M.D., Craigie I.D., Woodley S., Balmford A., Brooks T.M., Hockings M., Knights K., Mascia M.B., McRae L. & Burgess N.D. 2018. A global analysis of management capacity and ecological outcomes in terrestrial protected areas. *Conserv. Lett.* 11: 1–10.

Gray C.L., Hill S.L.L., Newbold T., Hudson L.N., Börger L., Contu S., Hoskins A.J., Ferrier S., Purvis A. & Scharlemann J.P.W. 2016. Local biodiversity is higher inside than outside terrestrial protected areas worldwide. *Nat. Commun.* 7: 12306. https://doi.org/10.1038/ncomms12306

Gregory R., Failing L., Harstone M., Long G., McDaniels T. & Ohlson D. 2012. *Structured decision making - A practical guide to environmental management choices.* Wiley-Blackwell, Chichester.

Grodzinska-Jurczak M. & Cent J. 2011. Expansion of nature conservation areas: Problems with natura 2000 implementation in Poland? *Environ. Manage.* 47: 11–27.

Groot R.S. de. 1987. Environmental functions as a unifying concept for ecology and economics. *Environmentalist* 7: 105–109.

Guerrero A.M., Knight A.T., Grantham H.S., Cowling R.M. & Wilson K.A. 2010. Predicting willingness-to-sell and its utility for assessing conservation opportunity for expanding protected area networks. *Conserv. Lett.* 3: 332–339.

Haapalehto T.O., Vasander H., Jauhiainen S., Tahvanainen T. & Kotiaho J.S. 2011. The Effects of Peatland Restoration on Water-Table Depth, Elemental Concentrations, and Vegetation: 10 Years of Changes. *Restor. Ecol.* 19: 587–598.

Haapanen M., Jansson G., Nielsen U.B., Braüner U., Steffenrem A. & Stener L-G. 2015. The status of tree breeding and its potential for improving biomass production: A review of breeding activities and genetic gains in Scandinavia and Finland. SkogForsk, Uppsala. Available from: https://urly.fi/1sHe (accessed February 2020).

Haight R.G. & Snyder S.A. 2009. Integer programming methods for reserve selection and design. In: Moilanen A., Wilson K.A. & Possingham H.P. (eds.), *Spatial Conservation Prioritization - Quantitative Methods and Computational Tools*, Oxford University Press, New York, pp. 43–57.

Haila Y. & Jokinen P. 2001. *Ympäristöpolitiikka - Mikä ympäristö, kenen politiikka*. Vastapaino, Tampere.

Hanley N., Banerjee S., Lennox G.D. & Armsworth P.R. 2012. How should we incentivize private landowners to 'produce' more biodiversity? *Oxford Rev. Econ. Pol.* 28: 93–113.

Hardin G. 1968. The Tragedy of the Commons. *Science* 162: 1243–1248.

Hase A. Von, Rouget M. & Cowling R.M. 2010. Evaluating private land conservation in the cape lowlands, South Africa. *Conserv. Biol.* 24: 1182–1189.

Henttonen H.M., Nöjd P. & Mäkinen H. 2017. Environment-induced growth changes in the Finnish forests during 1971–2010 – An analysis based on National Forest Inventory. *Forest Ecol. Manag.* 386: 22–36.

Hiedanpää J. 2002. European-wide conservation versus local well-being: the reception of the Natura 2000 Reserve Network in Karvia, SW-Finland. *Landscape Urban Plan.* 61: 113–123.

Hiedanpää J. 2005. The edges of conflict and consensus: A case for creativity in regional forest policy in Southwest Finland. *Ecol. Econ.* 55: 485–498.

Hooper D.U. 2005. Effetcs of biodiversity on ecosystem functioning: A consensus of current knowledge. *Ecol. Monogr.* 75: 3–35.

Hulme P.E. 2014. Bridging the knowing-doing gap: Know-who, know-what, know-why, know-how and know-when. *J. Appl. Ecol.* 51: 1131–1136.

Hyvärinen E., Juslén A., Kemppainen E., Uddström A. & Liukko U-M. (eds.) 2019. The 2019 Red List of Finnish Species. Ympäristöministeriö & Suomen ympäristökeskus, Helsinki.

Hänninen H., Leppänen J., Ovaskainen V., Uusivuori J. & Viitala E.-J. 2017. Metsätalouden uusi kannustinjärjestelmä – teoriaa, käytäntöjä ja ehdotukset. Luonnonvara- ja biotalouden tutkimus 5/2017. Available from: https://urly.fi/1u2I (accessed March 2020).

IPBES. 2019. *Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. Díaz S., Settele J., Brondízio E.S., Ngo H.T., Guèze M., Agard J., Arneth A., Balvanera P., Brauman K.A., Butchart S.H.M., Chan K.M.A., Garibaldi L.A., Ichii K., Liu J., Subramanian S.M., Midgley G.F., Miloslavich P., Molnár Z., Obura D., Pfaff A., Polasky S., Purvis A., Razzaque J., Reyers B., Roy Chowdhury R., Shin Y.J., Visseren-Hamakers I.J., Willis K.J., & Zayas C.N. (eds.). IPBES secretariat, Bonn.

IPCC. 2018. Summary for Policymakers. In: Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty. Masson-Delmotte V., Zhai P., Pörtner H-O., Roberts D., Skea J., Shukla P.R., Pirani A., Moufouma-Okia W., Péan C., Pidcock R., Connors S., Matthews J.B.R., Chen Y., Zhou X., Gomis M.I., Lonnoy E., Maycock T., Tignor M. & Waterfield T. (eds.).

Jackson-Smith D., Kreuter U. & Krannich R.S. 2005. Understanding the multidimensionality of property rights orientations: Evidence from Utah and Texas Ranchers. *Soc. Nat. Resour.* 18: 587–610.

Jenkins C.N., Pimm S.L. & Joppa L.N. 2013. Global patterns of terrestrial vertebrate diversity and conservation. *PNAS* 110: E2602–E2610.

Jokinen M., Hujala T., Paloniemi R. & Vainio A. 2018. Private landowners and protected species: What sort of noncompliance should we be worried about? *Global Ecol. Conserv.* 15: e00407. https://doi.org/10.1016/j.gecco.2018.e00407

Jones K.R., Venter O., Fuller R.A., Allan J.R., Maxwell S.L., Negret P.J. & Watson J.E.M. 2018. One-third of global protected land is under intense human pressure. *Science* 360: 788–791.

Joppa L.N. & Pfaff A. 2009. High and Far: Biases in the Location of Protected Areas. *PLoS ONE* 4: e8273. doi:10.1371/journal.pone.0008273

Jungkunst H.F. & Fiedler S. 2007. Latitudinal differentiated water table control of carbon dioxide, methane and nitrous oxide fluxes from hydromorphic soils: feedbacks to climate change. *Glob. Change Biol.* 13: 2668–2683.

Kaakinen E., Kokko A., Aapala K., Autio O., Eurola S., Hotanen J-P., Kondelin H., Lindholm T., Nousiainen H., Rehell S., Ruuhijärvi R., Sallantaus T., Salminen P., Tahvanainen T., Tuominen S., Turunen T., Vasander H. & Virtanen K. 2018. Suot. In: Kontula T. & Raunio A. (eds.), *Suomen luontotyyppien uhanalaisuus 2018. Luontotyyppienpunainen kirja – Osa 1: Tulokset ja arvioinnin perusteet*. Suomen ympäristökeskus ja ympäristöministeriö, Helsinki, pp. 117–170.

Kabii T. & Horwitz P. 2006. A review of landholder motivations and determinants for participation in conservation covenanting programmes. *Environ. Conserv.* 33: 11–20.

Kamal S., Grodzińska-Jurczak M. & Brown G. 2015a. Conservation on private land: a review of global strategies with a proposed classification system. *J. Environ. Plann. Man.* 58: 576–597.

Kamal S., Kocór M. & Grodzińska-Jurczak M. 2015b. Conservation opportunity in biodiversity conservation on regulated private lands: Factors influencing landowners' attitude. *Environ. Sci. Policy* 54: 287–296.

Kaplan J.O., Krumhardt K.M. & Zimmermann N. 2009. The prehistoric and preindustrial deforestation of Europe. *Quaternary Sci. Rev.* 28: 3016–3034.

Kareksela S., Moilanen A., Tuominen S. & Kotiaho J.S. 2013. Use of Inverse Spatial Conservation Prioritization to Avoid Biological Diversity Loss Outside Protected Areas. *Conserv. Biol.* 27: 1294–1303.

Kareksela S., Moilanen A., Ristaniemi O., Välivaara R. & Kotiaho J.S. 2018. Exposing ecological and economic costs of the research-implementation gap and compromises in decision making. *Conserv. Biol.* 32: 9–17.

Kareksela S., Aapala K., Alanen A., Haapalehto T., Kotiaho J.S., Lehtomäki J., Leikola N., Mikkonen N., Moilanen A., Nieminen E., Tuominen S. & Virkkala R. 2020. Combining spatial prioritization and expert knowledge facilitates effectiveness of large-scale mire protection process in Finland. *Biol. Conserv.* 241. doi: 10.1016/j.biocon.2019.108324.2020

Knight A.T. & Cowling R.M. 2007. Embracing opportunism in the selection of priority conservation areas. *Conserv. Biol.* 21: 1124–1126.

Knight A.T., Cowling R.M., Rouget M., Balmford A., Lombard A.T. & Campbell B.M. 2008. Knowing but not doing: Selecting priority conservation areas and the research-implementation gap. *Conserv. Biol.* 22: 610–617.

Knight A.T., Grantham H.S., Smith R.J., McGregor G.K., Possingham H.P. & Cowling R.M. 2011. Land managers' willingness-to-sell defines conservation opportunity for protected area expansion. *Biol. Conserv.* 144: 2623–2630.

Knight R.L. 1999. Private lands: The neglected geography. *Conserv. Biol.* 13: 223–224.

Laine J., Vasander H. & Laiho R. 1995. Long-Term Effects of Water Level Drawdown on the Vegetation of Drained Pine Mires in Southern Finland. *J. Appl. Ecol.* 32: 785–802.

Lambin E.F. & Meyfroidt P. 2011. Global land use change, economic globalization, and the looming land scarcity. *PNAS* 108: 3465–3472.

Langpap C. 2006. Conservation of endangered species: Can incentives work for private landowners? *Ecol. Econ.* 57: 558–572.

Law for reserving resources to mire drainage for forestry purposes and for making other unproductive or low productive forests as productive ones 140/1928. Available only in Finnish at the law information service Edilex: https://www.edilex.fi/smur/19280140 (accessed in December 2019).

Lehtomäki J. & Moilanen A. 2013. Methods and workflow for spatial conservation prioritization using Zonation. *Environ. Modell. Softw.* 47: 128–137.

Lehtomäki J., Tomppo E., Kuokkanen P., Hanski I. & Moilanen A. 2009. Applying spatial conservation prioritization software and high-resolution GIS data to a national-scale study in forest conservation. *Forest Ecol. Manag.* 258: 2439–2449.

Leifeld J. & Menichetti L. 2018. The underappreciated potential of peatlands in global climate change mitigation strategies. *Nat. Commun.* 9: 1–7.

Lele S., Springate-Baginski O., Lakerveld R., Deb D. & Dash P. 2013. Ecosystem services: Origins, contributions, pitfalls, and alternatives. *Conserv. Soc.* 11: 343–358.

Lele S., Wilshusen P., Brockington D., Seidler R. & Bawa K. 2010. Beyond exclusion : alternative approaches to biodiversity conservation in the developing tropics. *Curr. Opin. Env. Sust.* 2: 94–100.

Lewis D.J., Plantinga A.J., Nelson E. & Polasky S. 2011. The efficiency of voluntary incentive policies for preventing biodiversity loss. *Resour. Energy Econ.* 33: 192–211.

Lindholm T. & Heikkilä R. 2006. Destruction of mires in Finland. In: Lindholm T. & Heikkilä R. (eds.), *Finland - land of mires*, The Finnish Environment 23 | 2006, Finnish Environment Institute, Helsinki, pp. 179–192.

Liu J., Hull V., Batistella M., DeFries R., Dietz T., Fu F., Hertel T.W., Cesar Izaurralde R., Lambin E.., Li S., Martinelli L.., McConnell W.., Moran E.., Naylor R., Ouyang Z., Polenske K.., Reenberg A., Miranda Rocha G. de, Simmons C.., Verburg P.., Vitousek P.., Zhang F. & Zhu C. 2013. Framing sustainability in a telecoupled world. *Ecol. Soc.* 18. http://dx.doi.org/10.5751/ ES-05873-180226

Lochhead L.E. 1994. Preserving The Brownies' Portion: A History Of Voluntary Nature Conservation Organisations In New Zealand 1888–1935. Doctoral thesis, Lincoln University, New Zealand, available from: https://hdl.handle.net/10182/1563 (accessed February 2020).

Lueck D. & Michael J.A. 2003. Preemptive Habitat Destruction Under the Endangered Species Act. *J. Law Econ.* XLVI: 27–60.

Maanavilja L., Aapala K., Haapalehto T., Kotiaho J.S. & Tuittila E. 2014. Impact of drainage and hydrological restoration on vegetation structure in boreal spruce swamp forests. *Forest Ecol. Manag.* 330: 115–125.

Mace G.M., Collar N.J., Gaston K.J., Hilton-Taylor C., Akçakaya H.R., Leader-Williams N., Milner-Gulland E.J. & Stuart S.N. 2008. Quantification of extinction risk: IUCN's system for classifying threatened species. *Conserv. Biol.* 22: 1424–1442.

Mace G.M., Norris K. & Fitter A.H. 2012. Biodiversity and ecosystem services: A multilayered relationship. *Trends Ecol. Evol.* 27: 19–26.

Mair L., Mill A.C., Robertson P.A., Rushton S.P., Shirley M.D.F., Rodriguez J.P. & McGowan P.J.K. 2018. The contribution of scientific research to conservation planning. *Biol. Conserv.* 223: 82–96.

Malthus T. 1798. *An Essay on the Principle of Population*. Electronic Scholarly Publishing Project 1998. Available from: http://www.esp.org/books/malthus/population/malthus.pdf (accessed December 2019).

Margules C.R. & Pressey R.L. 2000. Systematic conservation planning. *Nature* 405: 243–253.

Mazor T., Giakoumi S., Kark S. & Possingham H.P. 2014. Large-scale conservation planning in a multinational marine environment: Cost matters. *Ecol. Appl.* 24: 1115–1130.

McShane T.O., Hirsch P.D., Trung T.C., Songorwa A.N., Kinzig A., Monteferri B., Mutekanga D., Thang H. Van, Dammert J.L., Pulgar-Vidal M., Welch-Devine M., Peter Brosius J., Coppolillo P. & O'Connor S. 2011. Hard choices: Making trade-offs between biodiversity conservation and human well-being. *Biol. Conserv.* 144: 966–972.

Meadows D.H., Meadows D.L., Randers J. & Behrens W.W. 1972. *The Limits to Growth. A Report for the Club of Rome's Project on the Predicament of Mankind*. Universe Books, New York.

Mikkonen N. & Moilanen A. 2013. Identification of top priority areas and management landscapes from a national Natura 2000 network. *Environ. Sci. Policy* 27: 11–20.

Millennium Ecosystem Assessment. 2005. *Ecosystems and Human Well-being: Synthesis*. Island Press, Washington, DC.

Ministry of Agriculture and Forestry. 2011. Ehdotus soiden ja turvemaiden kestävän ja vastuullisen käytön ja suojelun kansalliseksi strategiaksi. Brief of the working group preparing the National Peatland Strategy 16.2.2011. Available from: https://urly.fi/1sIH (accessed February 2020).

Ministry of the Environment. 2016. Everyman's right - Legislation and practice. The brochure. Retrieved 28 January, 2020, from https://www.ym.fi/en-US/Latest_News/Publications/Everymans_right_in_Finland(4484)

Ministry of the Environment. 2020. Hankkeen asettaminen Helmi-elinympäristö-ohjelman valmistelemiseksi. Appointment letter 21.1.2020. Available from: https://www.ym.fi/helmi (accessed February 2020).

Moilanen A. 2007. Landscape Zonation, benefit functions and target-based planning: Unifying reserve selection strategies. *Biol. Conserv.* 134: 571–579.

Moilanen A. 2008. Two paths to a suboptimal solution - once more about optimality in reserve selection. *Biol. Conserv.* 141: 1919–1923.

Moilanen A. & Arponen A. 2011. Administrative regions in conservation: Balancing local priorities with regional to global preferences in spatial planning. *Biol. Conserv.* 144: 1719–1725.

Moilanen A. & Ball I. 2009. Heuristic and approximate optimization methods for spatial conservation prioritization. In: Moilanen A., Wilson K.A. & Possingham H.P. (eds.), *Spatial Conservation Prioritization - Quantitative Methods and Computational Tools*, Oxford University Press, New York, pp. 58–69.

Moilanen A., Kujala H. & Leathwick J.R. 2009a. The Zonation framework and software for conservation prioritization. In: Moilanen A., Wilson K.A. & Possingham H.P. (eds.), *Spatial Conservation Prioritization - Quantitative Methods and Computational Tools*, Oxford University Press, New York, pp. 196–210.

Moilanen A., Possingham H.P. & Polasky S. 2009b. A mathematical classification of conservation prioritization problems. In: Moilanen A., Wilson K.A. & Possingham H.P. (eds.), *Spatial Conservation Prioritization - Quantitative Methods and Computational Tools*, Oxford University Press, New York, pp. 28–42.

Moilanen A., Wilson K.A. & Possingham H.P. 2009c. *Spatial conservation prioritization. Quantitative methods and computational tools*. Oxford University Press, New York.

Moilanen M., Hytönen J., Hökkä H. & Ahtikoski A. 2015. Fertilization increased growth of Scots pine and financial performance of forest management in a drained peatland in Finland. *Silva Fenn.* 49: 1–18.

Moilanen A., Franco A.M.A., Early R.I., Fox R., Wintle B. & Thomas C.D. 2005. Prioritizing multiple-use landscapes for conservation: Methods for large multi-species planning problems. *Proc. R. Soc. B.* 272: 1885–1891.

Moilanen A., Pouzols F.M., Meller L., Veach V., Arponen A., Leppänen J. & Kujala H. 2014. *Zonation - spatial conservation planning methods and software. Version 4. User Manual.* Available from: https://urly.fi/1pHG (accessed January 2020).

Moilanen A., Anderson B.J., Eigenbrod F., Heinemeyer A., Roy D.B., Gillings S., Armsworth P.R., Gaston K.J. & Thomas C.D. 2011. Balancing alternative land uses in conservation prioritization. *Ecol. Appl.* 21: 1419–1426.

Moon K. & Blackman D. 2014. A Guide to Understanding Social Science Research for Natural Scientists. *Conserv. Biol.* 28: 1167–1177.

Moore T.R. & Knowles R. 1989. The influence of water table level on methane and carbon dioxide emissions from peatland soils. *Can. J. Soil Sci.* 38: 33–38.

Natural Resources Institute Finland. 2019. Statistics database. Age of forest stands on forest land. Available from: https://urly.fi/1mt3.

Nature Conservation Act 1096/1996. Finlex Data Bank. Available in Finnish and in Swedish from: https://www.finlex.fi/fi/laki/ajantasa/1996/19961096

Nelson E., Mendoza G., Regetz J., Polasky S., Tallis H., Cameron D.R., Chan K.M.A., Daily G.C., Goldstein J., Kareiva P.M., Lonsdorf E., Naidoo R., Ricketts T.H., Shaw M.R. 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Front. Ecol. Environ.* 7: 4–11.

Norton D. 2000. Conservation biology and private land: Shifting the focus. *Cons. Biol.* 14: 1221–1223.

Official Statistics of Finland 2011–2016. Ownership of Forest Land. e-publication. Helsinki: Natural Resources Institute Finland. Available from: https://stat.luke.fi/en/ownership-forest-land

Olive A. 2016. It is just not fair: The Endangered Species Act in the United States and Ontario. *Ecol. Soc.* 21. http:// dx.doi.org/10.5751/ES-08627-210313

Olive A. & Mccune J.L. 2017. Wonder, ignorance, and resistance: Landowners and the stewardship of endangered species. *J. Rural Stud.* 49: 13–22.

Ottichilo W.K., de Leeuw J. & Prins H.H.T. 2001. Population trends of resident wildebeest [Connochaetes taurinus hecki (Neumann)] and factors influencing them in the Masai Mara ecosystem, Kenya. *Biol. Conserv.* 97: 271–282.

Paloniemi R. & Tikka P.M. 2008. Ecological and social aspects of biodiversity conservation on private lands. *Environ. Sci. Policy* 11: 336–346.

Paloniemi R. & Vilja V. 2009. Changing ecological and cultural states and preferences of nature conservation policy : The case of nature values trade in South-Western Finland. *J. Rural Stud.* 25: 87–97.

Paloniemi R., Massa I. & Tikka P. 2006. Metsänomistajat ja virallinen luonnonsuojelu. *Maaseudun uusi aika* 3/2006: 5–20.

Palosuo V.J. 1979. MERA programmes in the Finnish forestry. *Acta For. Fenn.* 165: 1–62. Only abstract available in English. https://doi.org/10.14214/aff.7599

Parkhurst G.M., Shogren J.F., Bastian C., Kivi P., Donner J. & Smith R.B.W. 2002. Agglomeration bonus: An incentive mechanism to reunite fragmented habitat for biodiversity conservation. *Ecol. Econ.* 41: 305–328.

Pereira H.M., Leadley P.W., Proença V., Alkemade R., Scharlemann J.P.W., Fernandez-Manjarrés J.F., Araújo M.B., Balvanera P., Biggs R., Cheung W.W.L., Chini L., Cooper H.D., Gilman E.L., Guénette S., Hurtt G.C., Huntington H.P., Mace G.M., Oberdorff T., Revenga C., Rodrigues P., Scholes R.J., Sumaila U.R. & Walpole M. 2010. Scenarios for global biodiversity in the 21st century. *Science* 330: 1496–1501.

Perrings C., Folke C. & Maler K.G. 1992. The ecology and economics of biodiversity loss: the research agenda. *Ambio* 21: 201–211.

Pin L., Miettinen J., Chin S. & Ghazoul J. 2011. Remotely sensed evidence of tropical peatland conversion to oil palm. *PNAS* 108: 5127–5132.

Pourcq K. De, Thomas E., Arts B., Vranckx A., Léon-Sicard T. & Damme P. Van. 2017. Understanding and Resolving Conflict Between Local Communities and Conservation Authorities in Colombia. *World Dev.* 93: 125–135.

Pouzols F.M., Toivonen T., Di Minin E., Kukkala A.S., Kullberg P., Kuusterä J., Lehtomäki J., Tenkanen H., Verburg P.H. & Moilanen A. 2014. Global protected area expansion is compromised by projected land-use and parochialism. *Nature* 516: 383–386.

Powers R.P. & Jetz W. Global habitat loss and extinction risk of terrestrial vertebrates under future land-use-change scenarios. *Nat. Clim. Change* 9: 323–329.

Pressey R.L. & Nicholls A.O. 1989. Efficiency in conservation evaluation: Scoring versus iterative approaches. *Biol. Conserv.* 50: 199–218.

Raftery A.E., Li N., Ševčíková H., Gerland P. & Heilig G.K. 2012. Bayesian probabilistic population projections for all countries. *PNAS* 109: 13915–13921.

Rands M.R.W., Adams W.M., Bennun L., Butchart S.H.M., Clements A., Coomes D., Entwistle A., Hodge I., Kapos V., Scharlemann J.P.W., Sutherland W.J. & Vira B. 2010. Biodiversity conservation: Challenges beyond 2010. *Science* 329: 1298–1303.

Rassi P., Hyvärinen E., Juslen A. & Mannerkoski I. 2010. *The 2010 Red List of Finnish Species.* Available in Finnish (English summary) from: http://hdl.handle.net/10138/299501 (accessed January 2020).

Raunio A., Schulman A. & Kontula T. 2008. *Assessment of Threatened Habitat Types in Finland - Part 1: Results and basis for assessment.* Available in Finnish (English summary) from: http://hdl.handle.net/10138/37900 (accessed January 2020).

Rayfield B., Moilanen A. & Fortin M.J. 2009. Incorporating consumer-resource spatial interactions in reserve design. *Ecol. Model.* 220: 725–733.

Regina K., Budiman A., Greve M.H., Grønlund A., Kasimir Å., Lehtonen H., Petersen S.O., Smith P. & Wösten H. 2016. GHG mitigation of agricultural peatlands requires coherent policies. *Clim. Policy* 16: 522–541.

Robinson J.G. 1993. The Limits to Caring: Sustainable Living and the Loss of Biodiversity. *Conserv. Biol.* 7: 20–28.

Rockwood L.L. 2015. *Introduction to Population Ecology - Second Edition*. Wiley Blackwell, Oxford.

Ruuhijärvi R. 1978. Soidensuojelun perusohjelma - Basic Plan for Peatland Preservation in Finland. *Suo* 29: 1–10. Only Summary available in English. Available from: http://www.suo.fi/article/9472 (accessed February 2020).

Rydin H. & Jeglum J. 2006. The Biology of Peatlands. Oxford University Press, Oxford.

Salomaa A., Paloniemi R. & Ekroos A. 2018. The case of conflicting Finnish peatland management – Skewed representation of nature, participation and policy instruments. *J. Environ. Manage.* 223: 694–702.

Salomaa A., Paloniemi R., Hujala T., Rantala S., Arponen A. & Niemelä J. 2016. The use of knowledge in evidence-informed voluntary conservation of Finnish forests. *Forest Policy Econ.* 73: 90–98.

Sharafi S.M., Moilanen A., White M. & Burgman M. 2012. Integrating environmental gap analysis with spatial conservation prioritization: A case study from Victoria, Australia. *J. Environ. Manage.* 112: 240–251.

Simmons B.A., Law E.A., Marcos-Martinez R., Bryan B.A., McAlpine C. & Wilson K.A. 2018a. Spatial and temporal patterns of land clearing during policy change. *Land Use Policy* 75: 399–410.

Simmons B.A., Marcos-martinez R., Law E.A., Bryan B.A. & Wilson K.A. 2018b. Frequent policy uncertainty can negate the bene fi ts of forest conservation policy. *Environ. Sci. Policy* 89: 401–411.

Sinclair S.P., Milner-Gulland E.J., Smith R.J., McIntosh E.J., Possingham H.P., Vercammen A. & Knight A.T. 2018. The use, and usefulness, of spatial conservation prioritizations. *Conserv. Lett.* 11: e12459. https://doi.org/10.1111/conl.12459

Soares-Filho B., Moutinho P., Nepstad D., Anderson A., Rodrigues H., Garcia R., Dietzsch L., Merry F., Bowman M., Hissa L., Silvestrini R. & Maretti C. 2010. Role of Brazilian Amazon protected areas in climate change mitigation. *PNAS* 107: 10821–10826.

Sorice M.G., Oh C.-O., Gartner T., Snieckus M., Johnson R. & Donlan C.J. 2013. Increasing participation in incentive programs for biodiversity conservation. *Ecol. Appl.* 23: 1146–1155.

Soroye P., Newbold T. & Kerr J. 2020. Climate change contributes to widespread declines among bumble bees across continents. *Science* 367: 685–688.

Soulé M.E. 1985. What is conservation biology? *BioScience* 35: 727–734.

Steffen W., Broadgate W., Deutsch L., Gaffney O. & Ludwig C. 2015. The trajectory of the anthropocene: The great acceleration. *Anthropocene Rev.* 2: 81–98.

Steffen W., Rockström J., Richardson K., Lenton T.M., Folke C., Liverman D., Summerhayes C.P., Barnosky A.D., Cornell S.E., Crucifix M., Donges J.F., Fetzer I., Lade S.J., Scheffer M., Winkelmann R. & Schellnhuber H.J. 2018. Trajectories of the Earth System in the Anthropocene. *PNAS* 115: 8252–8259.

Stephens S. 2001. Visions and viability: How achievable is landscape conservation in Australia? *Ecol. Manag. Restor.* 2: 189–195.

Sutherland W.J., Pullin A.S., Dolman P.M. & Knight T.M. 2004. The need for evidence-based conservation. *Trends Ecol. Evol.* 19: 305–308.

Tahvanainen T. 2011. Abrupt ombrotrophication of a boreal aapa mire triggered by hydrological disturbance in the catchment. *J. Ecol.* 99: 404–415.

Taylor P.D., Fahrig L., Henein K. & Merriam G. 1993. Connectivity is a vital element of landscape structure. *Oikos* 68: 571–573.

Tilman D., Clark M., Williams D.R., Kimmel K., Polasky S. & Packer C. 2017. Future threats to biodiversity and pathways to their prevention. *Nature* 546: 73–81.

Tolvanen A., Juutinen A. & Svento R. 2013. Preferences of Local People for the Use of Peatlands: The Case of the Richest Peatland Region in Finland. *Ecol. Soc.* 18. http://dx.doi.org/10.5751/ ES-05496-180219

Trewavas A. 2002. Malthus foiled again and again. *Nature* 418: 668–670.

Tulloch V.J.D., Tulloch A.I.T., Visconti P., Halpern B.S., Watson J.E.M., Evans M.C., Auerbach N.A., Barnes M., Beger M., Chadès I., Giakoumi S., McDonald-Madden E., Murray N.J., Ringma J. & Possingham H.P. 2015. Why do We map threats? Linking threat mapping with actions to make better conservation decisions. *Front. Ecol. Environ.* 13: 91–99.

United Nations General Assembly. 1987. *Report of the world commission on environment and development: Our common future*. United Nations General Assembly, Norway.

Urry L.A., Cain M.L., Wasserman S.A., Minorsky P.V. & Reece J.B. 2017. *Campbell Biology, 11th Edition*. Pearson Education, Inc., New York.

Vasander H. 2006. The use of mires for agriculture and forestry. In: Lindholm T. & Heikkilä R. (eds.), *Finland - land of mires*, The Finnish Environment 23 | 2006, Finnish Environment Institute, Helsinki, pp. 173–178.

Vasander H., Tuittila E., Lode E., Lundin L., Ilomets M., Sallantaus T., Heikkilä R., Pitkänen M.-L. & Laine J. 2003. Status and restoration of peatlands in northern Europe. *Wetl. Ecol. Manag.* 11: 51–63.

Vedeld P., Jumane A., Wapalila G. & Songorwa A. 2012. Protected areas, poverty and conflicts. A livelihood case study of Mikumi National Park, Tanzania. *Forest Policy Econ.* 21: 20–31.

Virtanen E.A., Viitasalo M., Lappalainen J. & Moilanen A. 2018. Evaluation, Gap Analysis, and Potential Expansion of the Finnish Marine Protected Area Network. *Front. Mar. Sci.* 5: 1–19.

Vucetich J.A., Bruskotter J.T. & Nelson M.P. 2015. Evaluating whether nature's intrinsic value is an axiom of or anathema to conservation. *Conserv. Biol.* 00: 1–12.

Wanless S., Harris M.P., Redman P. & Speakman J.R. 2005. Low energy values of fish as a probable cause of a major seabird breeding failure in the North Sea. *Mar. Ecol. Prog. Ser.* 294: 1–8.

Westman W.E. 1977. How Much Are Nature 's Services Worth ? *Science* 197: 960–964.

Wilhere G.F. 2008. The How-Much-Is-Enough Myth. *Conserv. Biol.* 22: 514–517.

Williams P. 2001. Complementarity. In: Levin S.A. (eds.), *Encyclopedia of Biodiversity*, Academic Press, San Diego, pp. 813–829.

Wilson K.A., Evans M.C., Marco M. di, Green D.C., Boitani L., Possingham H.P., Chiozza F. & Rondinini C. 2011. Prioritizing conservation investments for mammal species globally. *Philos. T. R. Soc. B.* 366: 2670–2680.

Worm B., Barbier E.B., Beaumont N., Duffy J.E., Folke C., Halpern B.S., Jackson J.B.C., Lotze H.K., Micheli F., Palumbi S.R., Sala E., Selkoe K.A., Stachowicz J.J. & Watson R. 2006. Impacts of biodiversity loss on ocean ecosystem services. *Science* 314: 787–790.

Wright J.B. 1992. Land trusts in the USA. *Land Use Policy* 9: 83–86.

Young J.C., Searle K., Butler A., Simmons P., Watt A.D. & Jordan A. 2016. The role of trust in the resolution of conservation conflicts. *Biol. Conserv.* 195: 196–202.

Zhang D. 2004. Endangered Species and Timber Harvesting: The Case of Red-Cockaded Woodpeckers. *Econ. Inq.* 42: 150–165.

ORIGINAL PAPERS

I

COMBINING SPATIAL PRIORITIZATION AND EXPERT KNOWLEDGE FACILITATES EFFECTIVENESS OF LARGE-SCALE MIRE PROTECTION PROCESS IN FINLAND

by

Santtu Kareksela, Kaisu Aapala, Aulikki Alanen, Tuomas Haapalehto, Janne Kotiaho, Joona Lehtomäki, Niko Leikola, Ninni Mikkonen, Atte Moilanen, Eini Nieminen, Seppo Tuominen & Raimo Virkkala 2020

Biological Conservation 241: 108324.

https://doi.org/10.1016/j.biocon.2019.108324

Biological Conservation 241 (2020) 108324

Contents lists available at ScienceDirect

Biological Conservation

journal homepage: www.elsevier.com/locate/biocon

Combining spatial prioritization and expert knowledge facilitates effectiveness of large-scale mire protection process in Finland

S. Kareksela[a,b,*], K. Aapala[c], A. Alanen[d], T. Haapalehto[a], J.S. Kotiaho[b,e], J. Lehtomäki[f], N. Leikola[c], N. Mikkonen[a,c,f], A. Moilanen[g,h], E. Nieminen[b,e], S. Tuominen[c], R. Virkkala[c]

[a] Metsähallitus, Parks & Wildlife Finland, Finland
[b] School of Resource Wisdom, University of Jyväskylä, Finland
[c] Finnish Environment Institute, Biodiversity Centre, Finland
[d] Finnish Ministry of the Environment, Finland
[e] Department of Biological and Environmental Science, University of Jyväskylä, Finland
[f] Department of Biosciences, University of Helsinki, Finland
[g] Finnish Natural History Museum, University of Helsinki, Finland
[h] Department of Geosciences and Geography, University of Helsinki, Finland

ARTICLE INFO

Keywords:
Spatial prioritization
Trade-offs
Implementation
Expert knowledge
Restoration prioritization
Mires

ABSTRACT

Conservation resource allocation involves a complex set of considerations including species, habitats, connectivity, local to global biodiversity objectives, alternative protection and restoration actions, while requiring cost-efficiency and effective implementation. We present a national scale spatial conservation prioritization analysis for complementing the network of protected mires in Finland. We show how spatial prioritization coupled with regional targets and expert knowledge can facilitate structured decision-making. In our application, discussion between experts was structured around the prioritization model enabling integration of quantitative analysis with expert knowledge. The used approach balances requirements of many biodiversity features over large landscapes, while aiming at a cost-effective solution. As a special analytical feature, mire complexes were defined prior to prioritization to form hydrologically functional planning units, including also their drained parts that require restoration for the planning unit to remain or potentially increase in value. This enabled selection of mires where restoration effort is supporting and benefitting from the core mire areas of high conservation value. We found that a key to successful implementation was early on structured co-producing between analysts, mire experts, and decision-makers. This allowed effective multidirectional knowledge transfer and evaluation of trade-offs related to the focal conservation decisions. Quantitative trade-off information was seen especially helpful by the stakeholders to decide how to follow the analysis results. Overall, we illustrate a realistic and applicable spatial conservation prioritization case supporting real world conservation decision-making. The introduced approach can be applied globally to increase effectiveness of large-scale protection and management planning of the diverse wetland ecosystem complexes.

1. Introduction

As human-induced habitat loss, degradation, and fragmentation proceed, and when resources for conservation are limited, prioritization among potential measures and areas to protect biological diversity is needed (Margules and Pressey, 2000; Foley et al., 2005; Game et al., 2013; Sinclair et al., 2018). Systematically targeted conservation actions should be implemented to cost-effectively maintain species and habitats that are under pressure (Margules and Pressey, 2000; Ferrier and Wintle, 2009). These actions need prioritization, which has at least two dimensions. First, the analytical dimension is about effective utilization of knowledge, data, methods and tools (e.g. Game et al., 2013). The second dimension is making analyses operational (e.g. Knight et al., 2008, 2009). While the first dimension may appear logically challenging, the latter meets with cognitive, psychological and societal complexities (Gilbert, 2011; Toomey et al., 2016). We need to understand how the most can be gained from the analyses and conversely, how the analyses can be co-produced so that the results are both relevant for the problem at hand as well as perceived legitimate by those affected (Knight et al., 2009; Game et al., 2013; Young et al., 2014). Overall,

* Corresponding author at: Metsähallitus Parks & Wildlife, Finland.
E-mail address: santtu.kareksela@metsa.fi (S. Kareksela).

https://doi.org/10.1016/j.biocon.2019.108324
Received 1 April 2019; Received in revised form 26 October 2019; Accepted 28 October 2019
Available online 22 November 2019
0006-3207/

usefulness and implementation of systematic prioritization meets with complex ecological, societal and economic reality (Hirsch et al., 2010; Young et al., 2014; Paloniemi et al., 2017; Sinclair et al., 2018). Failing to fill the space between systematic analyses and their implementation to decision-making can lead to biases and opportunism in the decision-making process, decreasing the cost-effectiveness of the use of conservation resources (e.g. Game et al., 2013).

The relevance, legitimacy, credibility, and hence the overall usefulness of systematic conservation planning solutions is enhanced if the knowledge-implementation space is filled with genuine dialogue and co-producing involving analysis providers and decision-makers (Ferrier and Wintle, 2009; Young et al., 2014; Toomey et al., 2016; Bertuol-Garcia et al., 2017; Sinclair et al., 2018). Co-producing in the form of joint problem identification, formulation and investigating the solution and the related tradeoffs is strongly emphasized in the systematic conservation planning framework (Margules and Sarkar, 2007; Ferrier and Wintle, 2009; Knight et al., 2009; Kareksela et al., 2018). In addition to the more general operational model for conservation planning process, specific models exist especially for the implementation of results of conservation science and planning (Knight et al. 2006, 2010) along with e.g. a structured decision-making framework (Keeney and Raiffa, 1993; Gregory et al., 2012a) with practical examples (Gregory et al., 2012b; Guerrero et al., 2017).

While the importance of systematic conservation planning or spatial conservation prioritization analyses in providing solutions for wicked problems has been repeatedly demonstrated (Margules and Pressey, 2000; Pressey and Bottrill, 2008; Game et al., 2013), there are also other sources of information like local social-ecological considerations or expert knowledge that differ from the more systematic nature of spatial conservation prioritization approaches (Cowling et al., 2003; Drescher et al., 2013). Considering local stakeholders and expert knowledge is also valuable and often complementary and can influence decisions on social-ecological aspects not easily considered through e.g. more systematical spatial conservation planning approaches (Cowling et al., 2003; Drescher et al., 2013). Striking a balance between e.g. expert knowledge and quantitative analysis should be emphasized in decision-making processes (Cowling et al., 2003; Ferrier and Wintle, 2009) in order to achieve not only credibility but also relevance and legitimacy of the proposed solutions (Young et al., 2014). Systematic and structured utilization of expert knowledge is not self-evident or easy (Cowling et al., 2003), but it could be enhanced for example by integrating the use of expert knowledge with spatial prioritization analyses in a controlled way, e.g. by following the operational models of the systematic conservation planning and structured decision-making frameworks. The use of expert knowledge is also in a key role in designing the systematic analyses, emphasizing the need for structured multi-way knowledge transfer and co-production between the experts, analysis producers, and decision-makers.

Despite existing operational strategies (e.g. Margules and Sarkar, 2007; Knight et al., 2006, 2010; Gregory et al., 2012a) or more thematic approaches to increase the implementation success of the results of systematic analyses or prioritization knowledge in general (Hulme, 2014; Toomey et al., 2016; Young et al., 2014; Bertuol-Garcia et al., 2017), there still appears to be a shortage of practical examples on how conservation prioritization analyses and expert knowledge are integrated with on-the-ground decisions (Sinclair et al., 2018), without losing the effectiveness of either one of them. Here we present a spatial prioritization analysis to support decisions about complementing mire protection in Finland. We show how multiple information sources and a relatively complex complementary-based decision support analysis can be systematically integrated to the actual conservation decision-making process. Major contribution of this work is the effective use of the trade-off investigation (e.g. Kareksela et al., 2013, 2018; Kukkala and Moilanen, 2013), which is a key component in any structured decision-making process (Gregory et al., 2012a). We describe how information about prioritization trade-offs was used to fill the implementation space by helping to convey the analysis results to the decision-makers in an effective and user-friendly manner. We also provide a method to prioritize diverse mire complexes with restoration considerations. Mires as part of freshwater wetlands have a considerable impact on global biodiversity and ecosystem services and a high need for effective conservation actions (MEA, 2005). However, their prioritization as complex hydrological entities is still poorly reported.

2. Methods

2.1. Commissioning and aims of the work

The present prioritization work was commissioned by the working group set by the Finnish Ministry of the Environment to plan complementary expansion of the current network of protected mires in Finland (hereafter the complementary mire protection program, CMPP). The aim of the CMPP was to enhance the protection of a representative network of mires across different vegetation zones in Finland. The initiative for the CMPP and its protection target of approximately 100 000 ha was set in the government resolution following the Finnish peatland strategy (see e.g. Salomaa et al., 2018). CMPP covered the whole of Finland except for northern Lapland and the Åland Islands in the south-west (Fig. 1, Appendix Fig. A1 in Supplementary material). The program and the spatial prioritization analysis covered 1533 candidate mires (327 300 ha) and 3400 already protected mires (601 700 ha).

The working group comprised 14 stakeholders and experts, including experts of mire ecology, land-use planners, conservation scientists, and representatives of the environmental and forestry administration, the land owner's association and conservation NGOs. The working group reached a consensus that the added value of including a quantitative analysis to the decision-making process would be a complementarity-based evaluation accounting for many species and habitats. To include ecological connectivity in the evaluation was also seen important. The ecological model for the analysis (Fig. 2) was right from the start designed in close cooperation between the working group (represented here by the authors AA, KA, JSK) and experts in spatial prioritization (SK, AM, JL, NL, NM, TH, ST, RV). In addition to the described complementary based spatial prioritization analysis, the working group also created a scoring system for the candidate sites (including e.g. habitats, species, and naturalness of the mires) that was used to help the expert work. Together the prioritization analysis and the scoring formed a *Structured Decision Making* (SDM) process (e.g. Gregory et al., 2012a) where the problem was formulated, targeted effects of different data and analysis elements (Fig. 2) were heuristically defined, and, following the analysis, related trade-offs were systematically and quantitatively (when applicable) explored.

Decision-making process can have several phases before the areas chosen for example according to a prioritization analysis are set aside and protected in the landscape (Kareksela et al., 2018). Here we consider implementation as the process where the analyses results are integrated to the decision-making, i.e. to the process of choosing mire areas for protection before the process of actually setting the sites aside in the landscape. It should be noted that while the implementation in terms of decision-making process of selecting the sites for protection has been completed (Alanen and Aapala, 2015) the final implementation of the complementary expansion of the protected area network in the landscape is still partially to be carried out. A more detailed account of the protection program is given by Alanen and Aapala (2015) and its social-ecological context in Finland by Salomaa et al. (2018).

2.2. Data

Spatial data was based on a comprehensive field survey of the candidate sites, a pre-existing and highly detailed habitat database on protected areas, small water bodies from topographic database (streams

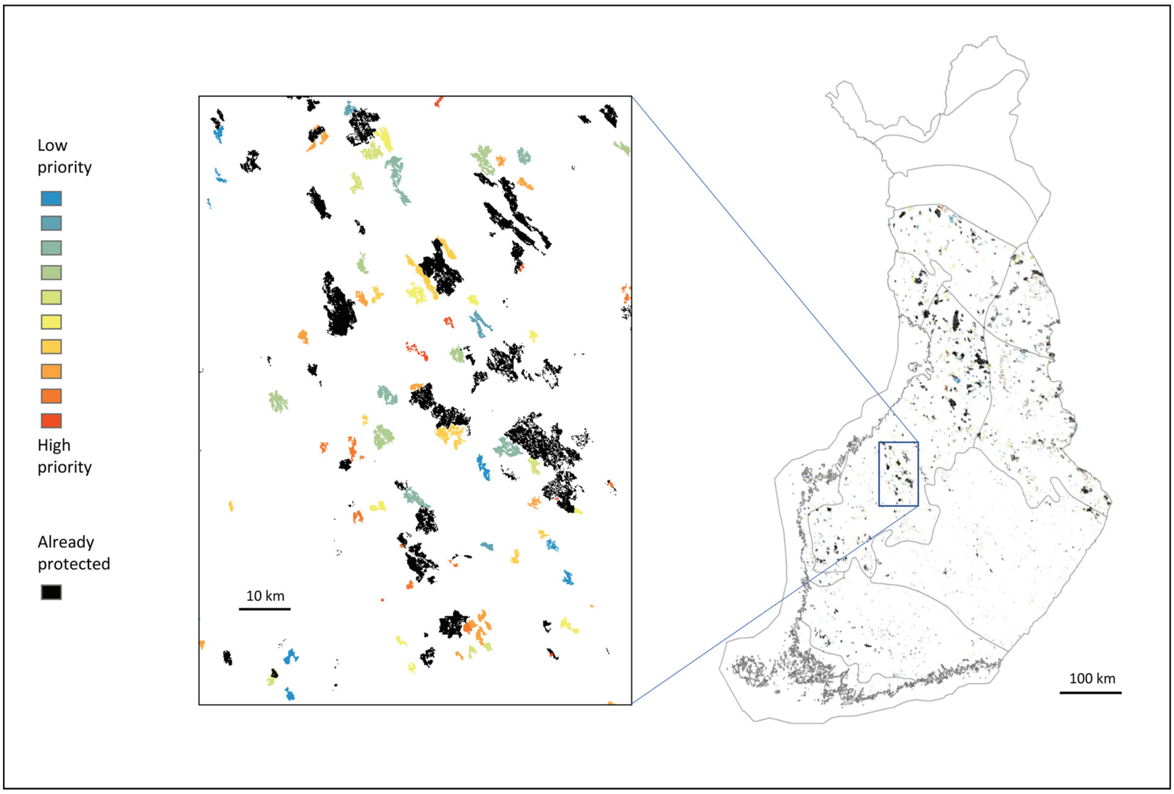


Fig. 1. Analysis area in Finland with administrative unit (forest vegetation zones) borders (see Appendix for further information) and a higher resolution map showing the priorities at the level of individual planning units (hydrological mire entities). Black areas represent the already protected mires and the colored areas from blue (low priority) to red (high priority) represent the prioritized candidate sites.

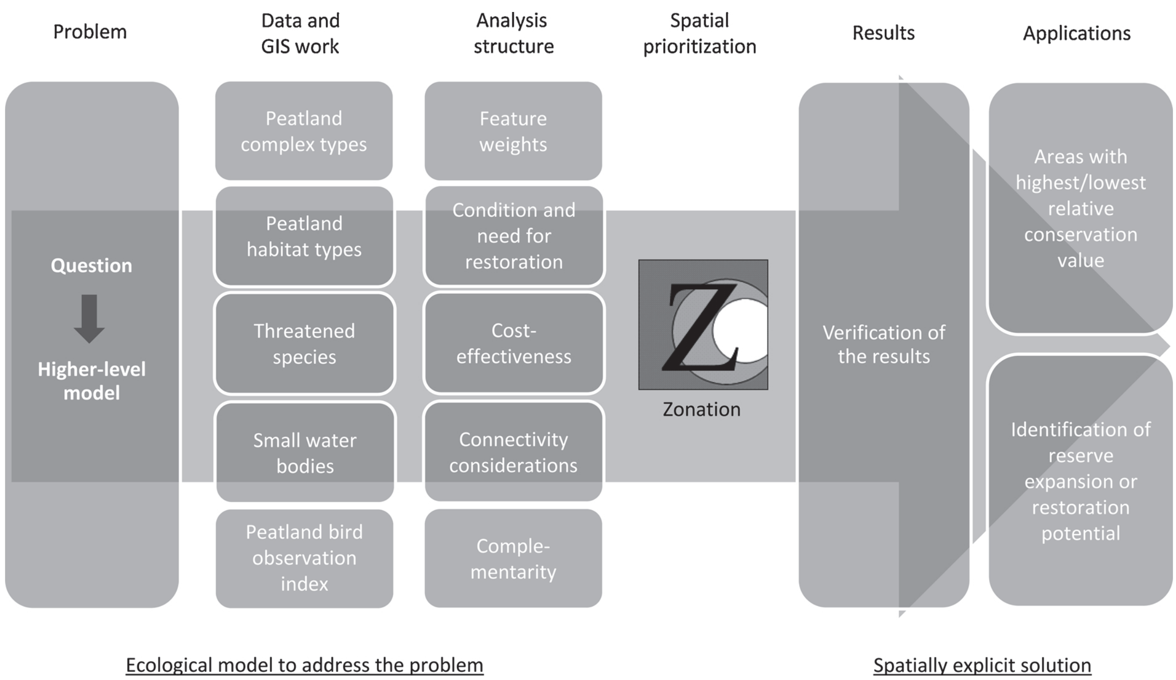


Fig. 2. Flowchart of the complementarity-based analysis implemented using the decision support tool Zonation. All the phases and use of the data elements and analysis approaches were co-planned together by the analysis producers and the mire experts, end-users and other stakeholders in the working group.

and ponds, National land survey of Finland), modelled likelihood for mire bird species territories (based on Breeding Bird Atlas), and species observation data from the Finnish threatened species database HERTTA. Altogether 91 spatial data layers were collated on geomorphological mire complexes (31 layers), mire habitats (39), threatened plants and mosses (3), small water bodies (1), and potential habitats for mire associated birds (17). As condition data for the mires, we used spatial data on ditches on peatlands from topographic database of National Land Survey of Finland. All spatial data were included in analysis as 50-meter resolution raster data layers. See Appendix in Supplementary material for more detailed information on the data.

2.3. Prioritization model and analysis structure

We performed the prioritization analysis using the freely available Zonation approach and software (Moilanen, 2007; Moilanen et al., 2005, 2014). Zonation is a spatial prioritization framework that can identify areas important for retaining habitat quality and connectivity simultaneously for all biodiversity features in the landscape, thereby indirectly aiming at retaining maximal population sizes and persistence of features (Lehtomäki and Moilanen, 2013; Moilanen et al., 2005, 2014). In the ecological sense, Zonation balances the (biodiversity) feature representation in terms of feature quality, amount and connectivity. Simultaneously, ecological considerations can be balanced against multiple direct costs, indirect costs and alternative land uses (Moilanen et al., 2011; Kareksela et al., 2013). Using an iterative process, Zonation produces a balanced priority ranking through the study landscape (Moilanen et al., 2005, 2014). The main outputs of Zonation are the priority rank map and (biodiversity) feature-specific information on what fraction of the original distribution of the feature can be covered (given the Zonation priority ranking) by protecting a given fraction of the landscape.

We constructed the analysis in stages so that we could investigate how different data and analysis approaches affected the priority of areas and that we could modify the analysis where needed. More information on the used analysis approaches can be found in the Appendix and Zonation manual (Moilanen et al., 2014). Concerning the spatial prioritization analysis applied in the present work, following methods were used (see Appendix for details in Supplementary material):

1 We weighted the biodiversity features in the analysis based on their Red List status across the whole planning area and more detailed regional Red List statuses (Raunio et al., 2008; Rassi et al., 2010).
2 We used a *hierarchical mask* layer to separate between (prioritize in sequential steps) present protected area network and candidate sites to identify which candidate sites best complement the already protected mire biodiversity (e.g. Mikkonen and Moilanen, 2013; Virtanen et al., 2018).
3 We did the analysis on *planning units* (groups of grid cells) to enable prioritization of the mire ecosystem complexes as hydrological entities (spatially defined by mire experts, Appendix, Fig. A3 in Supplementary material).
4 We applied a *condition layer* to de-emphasize areas where land-use pressure has led to loss of ecological condition and respectively to identify restoration need. Here the use of the condition layer prioritizes sites that include comparatively less damage from drainage (Kareksela et al., 2013, 2018). If partially damaged, the site will only receive high conservation priority in the analysis if its complementary biodiversity value outweighs the lowered condition. The potentially needed restoration actions at the top priority sites also support the persistence of mires with high complementary biodiversity value, thus also leading to prioritized use of resources for restoration.
5 We used *administrative unit analysis* to allow balancing of local and national scale rarity and weighing of the biodiversity features (Moilanen and Arponen, 2011). Heuristically expressed, this type of analysis simultaneously considers both regional and national priorities, leading to more spatially balanced distribution of top priority areas among the considered regions.
6 To emphasize the ecological connectivity of areas we applied *interaction connectivity* (Lehtomäki et al., 2009; Rayfield et al., 2009). This method increases the priority of candidates mires that have other ecologically high-quality areas nearby. Connectivity was emphasized between all mires/peatlands included in the analysis, with higher contributions counted from areas already protected.

2.4. Post processing and integration of expert knowledge

Zonation outputs the priority ranking as geospatial raster data, which can be visualized using any GIS software. In the present case, other main outputs relevant for the working group included a priority listing of the candidate mires (planning units) and information about the representation levels (coverage) of biodiversity features in the solution (sites chosen according to the Zonation analysis, hereafter: the solution). We used the feature-specific representation levels produced by Zonation (Moilanen et al., 2014) to investigate trade-offs between features or groups of features within and between solutions, when the final analysis was iteratively built. Next, we plotted the feature-specific representation levels to produce so-called performance curves to illustrate how the biodiversity coverage of a solution continuously improves when new areas are added into the PA network. Typically, gains are highest with the first additions and level off (saturate) when moving to lower priority additions (Fig. 3). This is because the first additions effectively implement gap-filling: they rapidly add coverage for many (comparatively) narrow-range features that have missing or low representation in the existing PA network (Sharafi et al., 2012; Virtanen et al., 2018). This information was useful for the working group in deciding the fraction of new sites (the solution) that should be chosen primarily based on the quantitative Zonation analysis compared to the area that could be primarily chosen based on e.g. the scoring and expert knowledge. We used ArcMap (ESRI 2014) to process the analysis outputs and for the spatial analysis of the results.

The co-production process was centered around the systematic prioritization analysis and carried out without a specific co-production method or model. In other words, building up the model for the prioritization analysis and the expert work for the scoring system (see above) served as a platform for structured decision-making. The stakeholders needed to agree on settings to address the formulated problem, e.g. by defining what elements (data, connectivity, regional priorities etc.) they wanted to include in the quantitative analysis, and how each element was to be emphasized. In addition, the multiphased analysis process (above) served to provide structure for the knowledge use. The co-production and integration of the analysis results was carried out in stages:

1) The stakeholders, mire experts, and analysis experts outlined the detailed targets that would best serve the goal of the program. (It should be noted that the used prioritization method does not require habitat or species-specific targets for biodiversity representation, but it instead aims to cumulative persistence of the most complementary biodiversity features).
2) Alongside with building the model the performance of the analysis was monitored (Figs. 3 and 4) as the key parameters were iterated.
3) The working group used the performance curves produced in the final analysis to define complementarity saturation (Fig. 3), which was then used to decide how far to follow the prioritization analysis solution (continuous ranking of the candidate mires, see Fig. 1 and above) and how much resources to leave for other prioritization principles, i.e. candidate site scoring and expert knowledge.
4) After deciding how much the analysis priority was followed, the candidate sites suggested according to the analysis results were also

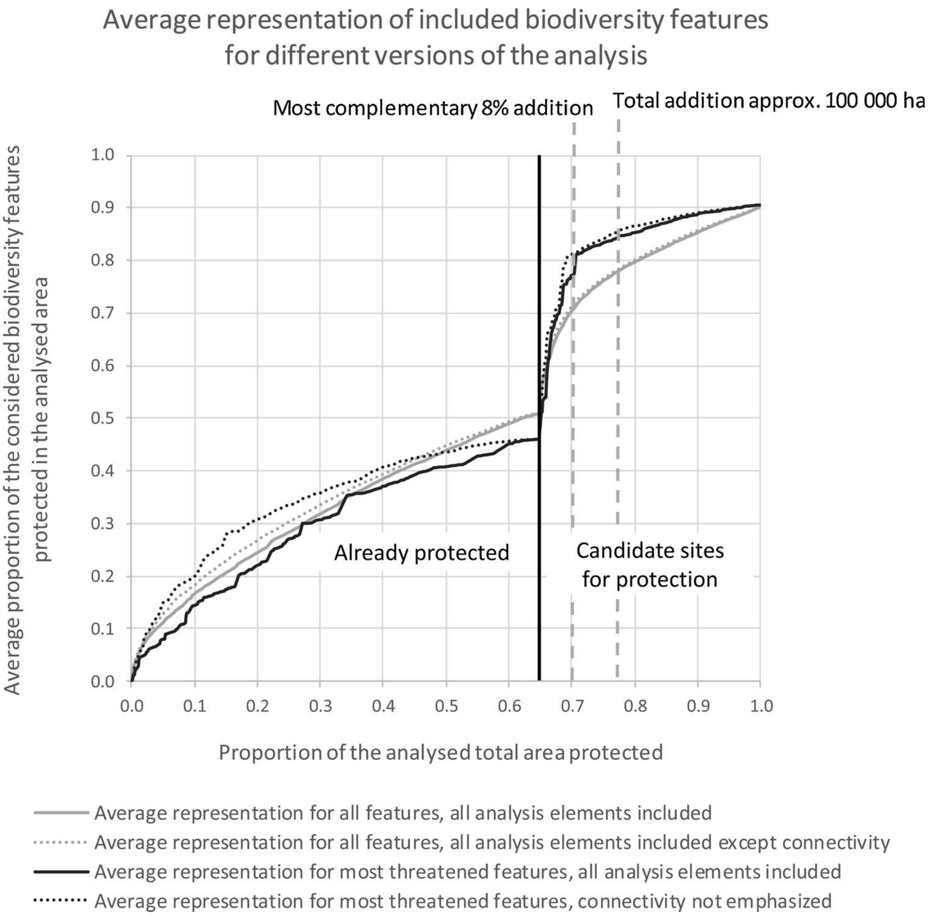


Fig. 3. Performance curves describe how the coverage of biodiversity features changes as function of area added into protection. The black vertical line indicates division between presently protected areas (left from line) and expansion areas (right). The steep rise of the curves to the right of the black line means that some species or habitats are missing or poorly represented within present protected areas (see also Fig. 4), but that the coverage of these features can be improved rapidly with additional sites, until representativeness increase starts to saturate. The dashed gray vertical line on the left shows the amount suggested to be chosen according to the Zonation analysis to ensure complementary solution (the solution of 8% addition to what is already protected), and the dashed gray vertical line on the right marks the total additional area suggested to be protected by the CMPP.

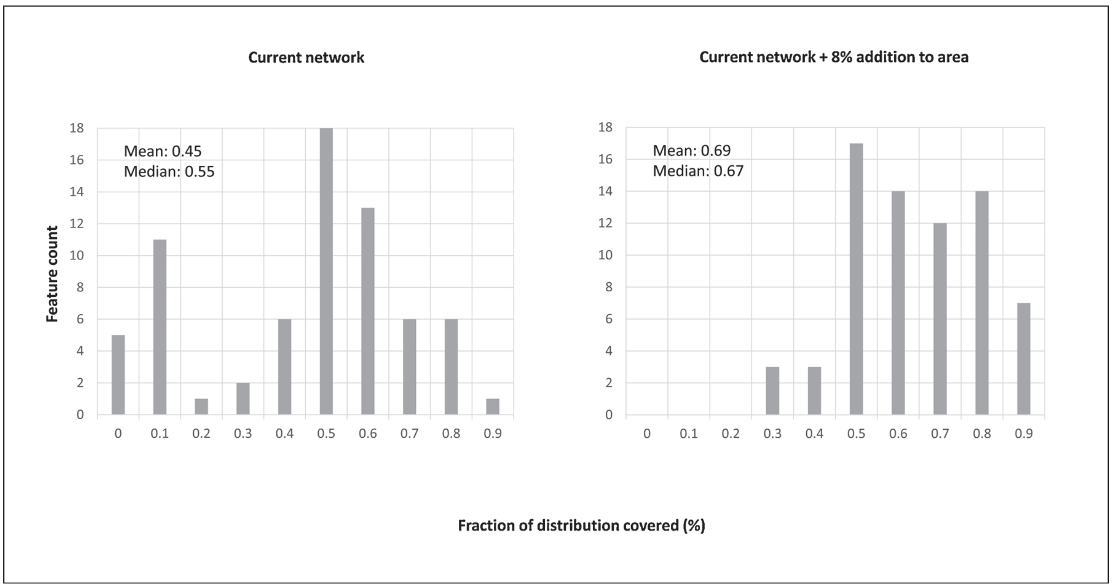


Fig. 4. Histogram of coverage of input biodiversity features (excluding modelled potential distributions for birds and small water bodies) relative to the features' abundances in the current protected areas and candidate areas combined (total analysis area). Comparison of the histograms for current network and with 8 % increase in its area, demonstrate the filling in the biodiversity gaps, i.e. allocating conservation resources for the features least well represented in the current network. Figures on the x-axis show the fraction of a features' analyzed abundance protected currently (left histogram) and in a +8 % situation (right histogram) and the y-axis shows the number of features that have that fraction of its abundance protected. Note that the +8 % represents the mire sites chosen for protection according to Zonation analysis solution (best 8 % addition to the protected landscape) and the final increase in biodiversity representation is higher than shown in the figure, because the addition presented here was approximately 1/3 of the total area chosen for protection. Thus, the presented figure is to demonstrate the "gap-filling effect" of complementarity-based analysis used here.

qualitatively examined by the working group to, for example, spot major mistakes in data and to make sure that no “strange choices” existed.

5) The actual site selection was then carried out hierarchically, first choosing areas according to the spatial prioritization analysis that best complement the existing protection network, and then complementing this with highest scoring areas within each administrative unit, based on the used scoring method, up to the approx. 100,000 ha total target.

To investigate how restoration need was reflected in the solution we compared the area of the candidate mire sites that needed restoration between all the candidate sites and the candidate sites chosen according to our analysis (the solution). We also calculated the proportion of the area needing restoration that was on sites that hosted the most threatened peatland complexes and habitats and compared this between all the candidate sites and the sites suggested for protection in the solution.

3. Results

3.1. Balanced solution for the complementary expansion

The ecological value of the mire protection area network increased as a function of gradual addition of candidate areas, as shown by the performance curves (Fig. 3). The highest ranked 5% of the analysis area (candidate sites), which corresponds to an approximately 8% increase of protected mire area, would achieve on average a 39% relative increase in the conservation coverage of biodiversity features included in analysis. High cost-effectiveness is primarily achieved via additions for narrow-range features that have missing or low representation in the existing PA network. Coverage of the most threatened features (mire complexes and habitats) improved significantly more than coverage for all features, a 68% relative improvement for the 8% area increase (Fig. 3).

It is not only conservation coverage of biodiversity features that is increased in this process, but also the balance of coverage across species improves. Because the analysis is based on complementarity, well balanced additions into the network also fix gaps in the ecological coverage of the network (Fig. 4). As shown in Fig. 4, the analysis was effective in rising the representation levels of the features with lowest representations at the current network for protected mires. Hence, the analysis can be considered a cost-effective solution for complementing the network for protected mires in Finland.

The performance curves were also used to investigate a potential trade-off between direct biodiversity coverage and spatial connectedness (Fig. 3). They show that additional connectivity consideration did not have a significant negative effect on the static representation of biodiversity in the solution, although it did have an elevated effect for the highest weighted eight features (Fig. 3). This relatively small compromise was nevertheless considered to hold net positive potential for the long-term persistence of biodiversity.

We were also able to achieve a relatively balanced distribution of top priority mires in the solution over the administrative units. Because the priorities (expressed with feature weights) for certain habitats were higher in the south (Appendix, Table A2 in Supplementary material) and the overall drainage-based degradation of mires is significantly higher in the southern half of Finland, the solution also emphasized more the southern regions, although all the regions included mires in the solution (see Appendix, Fig. A2 in Supplementary material for map of the administrative units and solutions spatial distribution). The balance of the distribution of mires over the planning area was further complemented with the protection choices made by the working group.

3.2. Spatial allocation of restoration resources

The area of candidate mires in need of restoration was 29% of the total of candidate sites and 20% of the sites suggested for protection in the solution (the 8% addition to protection). On average 55% of the area on all the candidate mires that needed restoration was on sites that host one or more highest-weighted mire complexes and/or habitats (see methods and Appendix in Supplementary material). Of the candidate mires in the solution chosen according to the Zonation analysis, 78% of the area needing restoration was on sites that host these top features. This means that the analysis was effective in choosing areas in good condition, i.e. areas with lower need of resources for restoration, where the still remaining need for restoration efforts was strongly associated to areas with high priority habitats, which increases the cost-effectiveness of potential future restoration efforts on these sites.

3.3. Integration to decision-making

The results described above were presented to the working group, which identified the most cost-effective set of areas that fill gaps in conservation coverage efficiently. Using the graphical illustration of the heuristically defined “saturation of representativeness increase”, the analysis providers (SK, AM, JL, NM, NL, RV, ST, TH) were able to produce a recommendation about the number and identity of sites that should preferably be chosen to retain most benefits of the complementary solution. This was approximately 1/3 of the area that could be chosen within the program’s area-based target limitation (Fig. 3). The remaining 2/3 of the targeted additional area for protection could then be chosen according to the highest regional scoring points. This approach, along with the connectivity, restoration, and regional considerations (see above) was welcomed and strongly supported by the working group, and it led to successful integration of the analysis results into the decision-making process.

The working group checked prioritization results for top sites to correct any false expectations of ecological value that might have arisen due to problems with data or the ecological model of conservation value. Sites were excluded mainly for practical reasons. For instance, some of the very small or recently drained sites were replaced with more representative candidate sites.

4. Discussion

We were able to produce a cost-effective solution for the complementary network of protected mires in Finland. The analysis results were also successful in facilitating the decision-making process. Through the spatial prioritization analysis, the stakeholder group was able to address a complicated problem involving rarity and Red List status of different biodiversity features, connectivity of areas, and variable restoration need of sites. Even with comparatively small addition to protected area (8%), it was possible to produce a very high increase (20%) in the representation of biodiversity in the PA network. This demonstrates the utility of a systematic analysis in a structured decision-making process, transforming the expert knowledge and stakeholders’ goals into effective conservation outcomes (Margules and Pressey, 2000; Ferrier and Wintle, 2009; Game et al., 2013). The resulting quantitative information of the trade-offs was used to facilitate integration of prioritization results with external decision criteria and expert opinion. We provided information about the characteristics of alternative solutions, allowing well-informed participation in decision-making by people less involved in the prioritization analysis itself.

The analysis presented here was successful both technically and, in its implementation characteristics. The working group was satisfied with clear presentation of the results and how the analysis facilitated the thought processes of the planning group. In other words, the analysis was able to act as a platform for structural decision-making (Gregory et al., 2012a) where the problem formulation and goals for the analysis model were co-produced among the stakeholders and experts. Systematic analysis also provided quantitative comparison of trade-offs, which facilitated the evaluation of the spatial prioritization analysis results and provided more general information on the trade-offs related to the focal conservation task. The role of this trade-off information was imperative in how the prioritization analysis results were implemented: the trade-off investigation offered a way to inspect how much to follow the analysis results and how much to leave freedom for resource allocation based on other considerations while still meeting the complementarity targets. This was seen a very useful way by the working group to decide what way and how much to follow the analysis results.

We emphasize the importance of expert knowledge in building the ecological model of conservation value and the analyses it enables. Not only would it be very difficult to build such a model without expert knowledge but engaging with experts and other stakeholders also seems to increase the chances of the results being more relevant for the decisions at hand as well as being perceived more legitimate (i.e. more acceptable). Integration of analysis results with expert knowledge also made the use of the expert knowledge more analytical (Drescher et al., 2013; Drescher and Edwards, 2018). All parties involved felt that the rather fluent multidirectional transfer of knowledge resulting from the co-production made a significant difference in how the prioritization analysis was used by the working group to make decisions about candidate sites. We believe this as a result further strengthens the knowledge on the benefits of multidirectional information transfer previously documented in the literature (e.g. Young et al., 2014; Toomey et al., 2016; Bertuol-Garcia et al., 2017). All in all, the structured analysis process with comprehensive involvement of experts and stakeholders seemed very efficient in filling the implementation space with cooperation, analytical information, and knowledge.

Consideration of the restoration needs and hydrological entities in the focal case is a special example of the help systematic spatial analyses can bring to large scale mire conservation globally, complementing the existing methods for conservation planning of wetlands (e.g. Choulak et al., 2019; Reis et al., 2019). Through the analysis methods of *planning units* and *condition layer* we were able to achieve a systematic way to balance the restoration needs with the biodiversity value of the candidate sites as hydrological units (Appendix, Fig. A3 in Supplementary material). A mire site was chosen into the set of complementary areas (the solution) despite it having lowered condition due to drainage, if its biodiversity value for the solution outbalanced its low condition. In addition, at the mires included in the solution that have lowered condition the need for restoration is linked to core areas of relatively high biodiversity value among the candidate mires, meaning that the restoration of these parts would provide hydrological support for valuable core areas while the core areas act as species sources for the restored parts. This is also a more general example of complexity arising from a need to do conservation decisions with respect to larger entities, based on for example hydrological connectedness. This is likely to apply to many wetland protection projects (Choulak et al., 2019; Reis et al., 2019), in addition to the mires in the focal case. However, although balancing condition with complementary representation of multiple biodiversity features, this was a rather simple consideration of restoration need. While more advanced optimization of restoration and management effects over multiple ecosystem types are likely to be needed in many other cases, it should be noted that they also considerably increase the analysis complexity (Possingham et al., 2009, 2015; Shoo et al., 2017). The introduced approach for mire conservation planning and for the use of trade-off information should have international interest, considering the global need for wetland conservation (MEA, 2005). However, the presented approach (as all large-scale planning) is dependent on the availability of reliable quantitative data, that is required for the analyses and is a corner stone of any trade-off evaluation (e.g. Kareksela et al., 2018; Kujala et al., 2018).

Here the data on land acquisition and restoration costs were not fully available at the time of the analysis. Missing the data on economic costs, we used area as a proxy for the cost to be minimized. This has a trade-off of its own by concentrating the solution more on the “hot spots” (high gain with small area) and lowering the probability for potential land-use conflicts in the future (minimizing needed total area) but not minimizing the actual economic resourcing from the society. If full cost information had been available, it could have influenced the relative priority of areas. It should be noted however, that costs did not restrict the protection program, but the societal target was more area-based (the 100,000 ha). As such the analysis could more efficiently fill its role as an information source for cost-effective solution to satisfy the complementarity goal by having the same limiting factor (i.e. area) as the whole decision-making process.

As usual, we lack a reference to be able to say how the results would have been used if the analysis had not been carefully co-produced within the working group. Ultimately, it is nearly impossible to know the impacts of alternative choices and analysis options (Sutherland et al., 2004; Sinclair et al., 2018). Following this problematization, it is difficult to speculate how well for example the structured decision-making process was carried out here, or how much better the prioritization analysis process or results could have been articulated to the stakeholders in the co-production and decision-making phases. Even so, quantitative evaluation of the trade-offs seemed to provide an excellent way to integrate expert knowledge, to evaluate alternative solutions, and to deliver results to decision-makers. In the end, a broad suite of factors was successfully converted into decisions about sites chosen for the protection program.

To conclude, in addition to providing a globally relevant method for effective large scale mire conservation, we were able to identify two major factors helping to fill in the implementation space between analyses and decision-making in a broad context. First, early on structured co-production between analysis experts, ecological experts, and the decision makers facilitates a successful analysis process closely linked to the actual decision-making. Second, systematic use of relevant trade-off information and a multidirectional benchmark process improves a balanced use of multiple information sources. Together, these conclusions strengthen observations made earlier (Hulme, 2014; Toomey et al., 2016; Young et al., 2014; Kareksela et al., 2018): open minds, open atmospheres, and open discussions are keys to successful cost-effective conservation.

Acknowledgements

We wish to thank all the members of the working group of the complementary mire protection program for the open-minded atmosphere and help in developing the presented analysis. We thank Pekka Salminen, Eero Kaakinen, Iida Kämäri, and Tuomas Lahti for their invaluable work for gathering and processing the spatial data. We also wish to thank the Finnish Ministry of the Environmentfor funding “MetZo II” and “SUMI” (Evaluating the protected area network in the chamgimg climate) -projects.

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.biocon.2019.108324.

References

Alanen, A., Aapala, K. (Eds.), 2015. Proposal of the Mire Conservation Group for Supplemental Mire Conservation, vol 26 Reports of the Ministry of the Environment pp. 175.

Bertuol-Garcia, D., Morsello, C.N., El-Hani, C., Pardini, R., 2017. A conceptual framework for understanding the perspectives on the causes of the science-implementation gap in ecology and conservation. Biol. Rev. https://doi.org/10.1111/brv.12385.

Choulak, M., Marage, D., Gisbert, M., Paris, M., Meinard, Y., 2019. A meta-decision-analysis approach to structure operational and legitime environmental policies - with an application to wetland prioritization. Sci. Tot. Environ. 655, 384–394.

Cowling, R.M., Pressey, R.L., Sims-Castley, R., Le Roux, A., Baard, E., Burgers, C.J., Palmer, G., 2003. The expert or the algorithm? - Comparison of priority conservation areas in the Cape Floristic Region identified by park managers and reserve selection software. Biol. Conserv. 112, 147–167. https://doi.org/10.1016/S0006-3207(02)00397-X.

Drescher, M., Perera, A.H., Johnson, C.J., Buse, L.J., Drew, C.A., Burgman, M.A., 2013. Toward rigorous use of expert knowledge in ecological research. Ecosphere 4, 1–26. https://doi.org/10.1890/ES12-00415.1.

Drescher, M., Edwards, R.C., 2018. A systematic review of transparency in the methods of expert knowledge use. J. Appl. Ecol. 56, 436–449.

Ferrier, S., Wintle, B.A., 2009. Quantitative approaches to spatial conservation prioritization: matching the solution to the need. In: Moilanen, A., Wilson, K.A., Possingham, H.P. (Eds.), Spatial Conservation Prioritization: Quantitative Methods and Computational Tools. Oxford University Press, Oxford, pp. 1–15.

Foley, J.A., Barford, C., Coe, M.T., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Ramankutty, N., DeFries, R., Asner, G.P., Bonan, G., Carpenter, S.R., Chapin, F.S., Daily, G.C., Prentice, I.C., Snyder, P.K., 2005. Global consequences of land use. Science 309, 570–574. https://doi.org/10.1126/science.1111772.

Game, E.T., Kareiva, P., Possingham, H.P., 2013. Six common mistakes in conservation priority setting. Conserv. Biol. 27, 480–485. https://doi.org/10.1111/cobi.12051.

Gilbert, D., 2011. Buried by bad decisions. Nature 474, 275–277.

Gregory, R., Failing, L., Harstone, M., Long, G., Mcdaniels, T., Ohlson, D., 2012a. Structured Decision Making: a Practical Guide to Environmental Management Choices. John Wiley & Sons, Ltd, Chichester, UK.

Gregory, R., Long, G., Colligan, M., Geiger, J.G., Laser, M., 2012b. When experts disagree (and better science won't help much): using structured deliberations to support endangered species recovery planning. J. Environ. Manage. 105, 30–43.

Guerrero, A.M., Shoo, L., Iacona, G., Standish, R.J., Catterall, C.P., Rumpff, L., de Bie, K., White, Z., Matzek, V., Wilson, K.A., 2017. Using structured decision-making to set restoration objectives when multiple values and preferences exist. Rest. Ecol 25, 858–865.

Hirsch, P.D., Brosius, J.P., Dammert, J.L., Adams, W.M., Bariola, N., Zia, A., 2010. Acknowledging conservation trade-offs and embracing complexity. Conserv. Biol. 25, 259–264. https://doi.org/10.1111/j.1523-1739.2010.01608.x.

Hulme, P.E., 2014. Bridging the knowing-doing gap: know-who, know-what, know-why, know-how and know-when. J. Appl. Ecol. 51, 1131–1136. https://doi.org/10.1111/1365-2664.12321.

Kareksela, S., Moilanen, A., Ristaniemi, O., Välivaara, R., Kotiaho, J.S., 2018. Exposing ecological and economic costs of the research-implementation gap and compromises in decision making. Conserv. Biol. 32, 9–17. https://doi.org/10.1111/cobi.13054.

Kareksela, S., Moilanen, A., Tuominen, S., Kotiaho, J.S., 2013. Use of inverse spatial conservation prioritization to avoid biological diversity loss outside protected areas. Conserv. Biol. 27, 1294–1303. https://doi.org/10.1111/cobi.12146.

Keeney, R.L., Raiffa, H., 1993. Decisions With Multiple Objectives: Preferences and Value Trade-offs. Cambridge University Press, Cambridge, UK.

Knight, A.T., Cowling, R.M., Rouget, M., Balmford, A., Lombard, A.T., Campbell, B.M., 2008. Knowing but not doing: selecting priority conservation areas and the research-implementation gap. Conserv. Biol. 22, 610–617.

Knight, A.T., Cowling, R.M., Possingham, H.P., Wilson, K.A., 2009. From theory to practice: designing and situating spatial prioritization approaches to better implement conservation action. In: Moilanen, A., Wilson, K.A., Possingham, H.P. (Eds.), Spatial Conservation Prioritization: Quantitative Methods and Computational Tools. Oxford University Press, Oxford, pp. 249–258.

Knight, A.T., Cowling, R.M., Boshoff, A.F., Wilson, S.L., Pierce, S.M., 2010. Walking in STEP: lessons for linking spatial prioritisations to implementation strategies. Biol. Conserv. 144, 202–211. https://doi.org/10.1016/j.biocon.2010.08.017.

Knight, A.T., Cowling, R.M., Campbell, B.M., 2006. An operational model for implementing conservation action. Conserv. Biol. 20, 408–419. https://doi.org/10.1111/j.1523-1739.2006.00305.x.

Kujala, H., Lahoz-Monfort, J.J., Elith, J., Moilanen, A., 2018. Not all data are equal: influence of data type and amount in spatial conservation prioritisation. Methods Ecol. Evol. 9, 2249–2261. https://doi.org/10.1111/2041-210X.13084.

Kukkala, A.S., Moilanen, A., 2013. Core concepts of spatial prioritisation in systematic conservation planning. Biol. Rev. 88, 443–464. https://doi.org/10.1111/brv.12008.

Lehtomäki, J., Moilanen, A., 2013. Methods and workflow for spatial conservation prioritization using Zonation. Environ. Model. Softw. 47, 128–137. https://doi.org/10.1016/j.envsoft.2013.05.001.

Lehtomäki, J., Tomppo, E., Kuokkanen, P., Hanski, I., Moilanen, A., 2009. Applying spatial conservation prioritization software and high-resolution GIS data to a national-scale study in forest conservation. For. Ecol. Manage. 258, 2439–2449. https://doi.org/10.1016/j.foreco.2009.08.026.

Margules, C.R., Pressey, R.L., 2000. Systematic conservation planning. Nature 405, 243–253.

Margules, C.R., Sarkar, S., 2007. Systematic Conservation Planning. Cambridge University Press, Cambridge, United Kingdom.

Mikkonen, N., Moilanen, A., 2013. Identification of top priority areas and management landscapes from a national Natura 2000 network. Environ. Sci. Pol. 27, 11–20.

MEA (Millennium Ecosystem Assessment), 2005. Ecosystems and Human Wellbeing: Wetlands and Water. Synthesis. Island Press, Washington, D.C.

Moilanen, A., 2007. Landscape Zonation, benefit functions and target-based planning: unifying reserve selection strategies. Biol. Conserv. 134, 571–579. https://doi.org/10.1016/j.biocon.2006.09.008.

Moilanen, A., Arponen, A., 2011. Administrative regions in conservation: balancing local priorities with regional to global preferences in spatial planning. Biol. Conserv. 144, 1719–1725. https://doi.org/10.1016/j.biocon.2011.03.007.

Moilanen, A., Franco, A.M.A., Early, R.I., Fox, R., Wintle, B., Thomas, C.D., 2005. Prioritizing multiple-use landscapes for conservation: methods for large multi-species planning problems. Proc. R. Soc. B Biol. Sci. 272, 1885–1891. https://doi.org/10.1098/rspb.2005.3164.

Moilanen, A., Anderson, B.J., Eigenbrod, F., Heinemeyer, A., Roy, D.B., Gillings, S., Armsworth, P.R., Gaston, K.J., Thomas, C.D., 2011. Balancing alternative land uses in conservation prioritization. Ecol. Appl. 21, 1419–1426. https://doi.org/10.1890/10-1865.1.

Moilanen, A., Pouzols, F.M., Meller, L., Veach, V., Arponen, A., Leppänen, J., Kujala, H., 2014. Zonation - spatial conservation planning methods and software. Version 4. User Manual. ISBN 978-952-10-9920-5 (PDF).

Paloniemi, R., Hujala, T., Rantala, S., Harlio, A., Salomaa, A., Primmer, E., Pynnönen, S., Arponen, A., 2017. Integrating social and ecological knowledge for targeting voluntary biodiversity conservation. Conserv. Lett. https://doi.org/10.1111/conl.12340.

Possingham, H.P., Moilanen, A., Wilson, K.A., 2009. Accounting for habitat dynamics in conservation planning. In: Moilanen, A., Wilson, K.A., Possingham, H.P. (Eds.), Spatial Conservation Prioritization: Quantitative Methods and Computational Tools. Oxford University Press, Oxford, pp. 135–144.

Possingham, H.P., Bode, M., Klein, C., 2015. Optimal conservation outcomes require both restoration and protection. PLoS Biol. https://doi.org/10.1371/journal.pbio.1002052.

Pressey, R.L., Bottrill, M.C., 2008. Opportunism, threats, and the evolution of systematic conservation planning. Conserv. Biol. 22, 1340–1345. https://doi.org/10.1111/j.1523-1739.2008.01032.x.

Rassi, P., Hyvärinen, E., Juslen, A., Mannerkoski, I., 2010. The 2010 Red List of Finnish Species (in Finnish, English summary). Available in. http://hdl.handle.net/10138/299501.

Raunio, A., Schulman, A., Kontula, T., 2008. Assessment of Threatened Habitat Types in Finland (in Finnish, English Summary). Available in. http://hdl.handle.net/10138/37900.

Rayfield, B., Moilanen, A., Fortin, M.J., 2009. Incorporating consumer-resource spatial interactions in reserve design. Ecol. Modell. 220, 725–733. https://doi.org/10.1016/j.ecolmodel.2008.11.016.

Reis, V., Hermoso, V., Hamilton, S.K., Bunn, S.E., Linke, S., 2019. Conservation planning for river-wetland mosaics: a flexible spatial approach to integrate floodplain and upstream catchment connectivity. Biol. Conserv. 236, 356–365.

Salomaa, A., Paloniemi, R., Ekroos, A., 2018. The case of conflicting Finnish peatland management – skewed representation of nature, participation and policy instruments. J. Environ. Manage. 223, 694–702. https://doi.org/10.1016/j.jenvman.2018.06.048.

Sharafi, S.M., Moilanen, A., White, M., Burgman, M., 2012. Integrating environmental gap analysis with spatial conservation prioritization: a case study from Victoria, Australia. J. Environ. Manage. 112, 240–251. https://doi.org/10.1016/j.jenvman.2012.07.010.

Shoo, L.P., Catterall, C.P., Nicol, S., Christian, R., Rhodes, J., Atkinson, P., Butler, D., Zhu, R., Wilson, K.A., 2017. Navigating complex decisions in restoration investment. Conserv. Lett. 10, 748–756.

Sinclair, S.P., Milner-Gulland, E.J., Smith, R.J., McIntosh, E.J., Possingham, H.P., Vercammen, A., Knight, A.T., 2018. The use, and usefulness, of spatial conservation prioritizations. Conserv. Lett. 11, e12459. https://doi.org/10.1111/conl.12459.

Sutherland, W.J., Pullin, A.S., Dolman, P.M., Knight, T.M., 2004. The need for evidence-based conservation. Trends Ecol. Evol. 19, 305–308. https://doi.org/10.1016/j.tree.2004.03.018.

Toomey, A.H., Knight, A.T., Barlow, J., 2016. Navigating the space between research and implementation in conservation. Conserv. Lett. 10, 619–625. https://doi.org/10.1111/conl.12315.

Virtanen, E.A., Viitasalo, M., Lappalainen, J., Moilanen, A., 2018. Evaluation, gap analysis, and potential expansion of the finnish marine protected area network. Front. Mar. Sci. 5, 1–19. https://doi.org/10.3389/fmars.2018.00402.

Young, J.C., Waylen, K.A., Sarkki, S., Albon, S., Bainbridge, I., Balian, E., Davidson, J., Edwards, D., Fairley, R., Margerison, C., McCracken, D., Owen, R., Quine, C.P., Stewart-Roper, C., Thompson, D., Tinch, R., van den Hove, S., Watt, A., 2014. Improving the science-policy dialogue to meet the challenges of biodiversity conservation: having conversations rather than talking at one-another. Biodivers. Conserv. 23, 387–404. https://doi.org/10.1007/s10531-013-0607-0.

II

NO EVIDENCE OF SYSTEMATIC PRE-EMPTIVE LOGGINGS AFTER NOTIFYING LANDOWNERS OF THEIR LANDS' CONSERVATION POTENTIAL

by

Eini Nieminen, Kalle Salovaara, Panu Halme & Janne Kotiaho 2020

Ambio, Early online.

https://doi.org/10.1007/s13280-020-01354-4

Ambio
https://doi.org/10.1007/s13280-020-01354-4

KUNGL. VETENSKAPS-AKADEMIEN
THE ROYAL SWEDISH ACADEMY OF SCIENCES

RESEARCH ARTICLE

No evidence of systematic pre-emptive loggings after notifying landowners of their lands' conservation potential

Eini Nieminen, Kalle Salovaara, Panu Halme, Janne Sakari Kotiaho

Received: 11 February 2020 / Revised: 2 May 2020 / Accepted: 2 June 2020

Abstract Landowners can intentionally impair biodiversity values occurring on their land to pre-empt biodiversity protection. This often leads to significant negative effects on biodiversity. We studied whether landowners in Finland engaged in pre-emptive loggings after they were notified that their wooded mires are candidate sites for a mire protection program. After the notification, harvesting rates of the candidate wooded mires were significantly lower compared to harvesting rates of similar but non-candidate wooded mires. Annual and monthly harvesting rates indicated that notifying landowners of the conservation potential did not launch systematic pre-emptive logging behavior. Nevertheless, part of the candidate wooded mires were logged, so some landowners place more weight on other values than the biodiversity ones. Pre-emptive behavior has been observed in other studies suggesting that many country- or system-specific factors such as cultural background or level of compensation can affect landowners' behavior.

Keywords Environmental policy · Forest conservation · Mire conservation · Panic clearing · Peatland · Private protected area

INTRODUCTION

Anthropogenic activity often degrades habitats resulting in reduction or even eradication of species' populations (Newbold et al. 2015). Therefore, restrictions in land use practices are an inevitable consequence of biodiversity protection. Land use restrictions are known to cause conflicts especially when conservation is based on command-and-control approaches such as the Endangered Species Act in the USA or the conservation program Natura 2000 in Europe. Both have been shaped with contradictions followed by for e.g., a lack of communication, information sharing, stakeholder involvement, and justice (e.g., Paavola 2003; Grodzinska-Jurczak and Cent 2011; Blicharska et al. 2016; Olive 2016). Landowners of areas hosting endangered species or habitats can have negative attitudes towards conservation actions for several reasons. For instance, land use restrictions to protect biodiversity can be considered as insulting property rights, being unfair actions, or causing economic harm (e.g., Jackson-Smith et al. 2005; Kabii and Horwitz 2006; Kamal et al. 2015; Blicharska et al. 2016; Olive 2016; Jokinen et al. 2018).

Command-and-control approaches can generate perverse incentives to intentionally destroy or damage species or habitats. Such behavior is here referred to as pre-emptive behavior. Landowners can manage their lands in ways that harm threatened species directly (Brook et al. 2003; Jokinen et al. 2018). Occurrences of threatened species can also lead to shortened rotation times of forest loggings or to an increased probability of forests becoming logged on nearby sites, thereby preventing the species from dispersing into new areas (Lueck and Michael 2003; Zhang 2004). Net reduction of forest area caused by pre-emptive behavior can also outcompete attempts to halt deforestation (Simmons et al. 2018a).

Net effects of command-and-control approaches on biodiversity have both positive and negative outcomes. The Endangered Species Act in the USA seems to protect species from extinctions and increases the likelihood of species' status to improve (Schwartz 2008), at least if species' listings to the Act are combined with sufficient

Electronic supplementary material The online version of this article (https://doi.org/10.1007/s13280-020-01354-4) contains supplementary material, which is available to authorized users.

Published online: 23 June 2020

species-specific funding (Ferraro et al. 2007; Gibbs and Currie 2012). Still, negative impacts of pre-emptive behavior on single species may be significant (Brook et al. 2003; Lueck and Michael 2003). In Australia, strict clearing bans based on the Vegetation Management Act have increased forest cover on some regions and forest types, but later changes and uncertainties in the implementation of the Act have caused pre-emptive deforestation and other perverse effects leading to a net loss of remnant forest patches (Simmons et al. 2018b).

While evidence about pre-emptive behavior comes mainly from the USA (e.g., Brook et al. 2003; Lueck and Michael 2003; Zhang 2004) and Australia (Simmons et al. 2018a, b), the topic is debated also in many other parts of the world. Increasingly more land is converted to human use and, consequently, the loss of biodiversity continues (Pereira et al. 2010; Lambin and Meyfroidt 2011; IPBES 2019). The role of private lands in biodiversity protection is increasing as these host significant proportions of distributions of many endangered species and habitats (e.g., Knight 1999; Norton 2000). It is likely that landowners in different countries may respond differently to the risk of land use restrictions caused by conservation actions. Such difference may arise due to e.g., previous environmental administrative practices or politics (Paloniemi and Vilja 2009), a cultural-specific relationship with nature and land (Silvasti 2003), or compensation practices (Byl 2019).

Establishing the European Union's Natura 2000 conservation network caused heavy opposition by local people throughout Europe (Alphandéry and Fortier 2001; Hiedanpää 2002; Paavola 2003; Grodzinska-Jurczak and Cent 2011). The opposition initiated a development towards voluntary-based conservation approaches during the 21st century (Keulartz 2009). Since then, voluntary nature protection has been a predominant tool in forest conservation in Finland, but for other habitat types similar administrative tools are still lacking (Council of State 2014; Paloniemi and Vilja 2009). In 2012, the Complementary Mire Protection Program (hereafter the CMPP) aiming to extend the national mire conservation network was politically agreed on to be based on the Nature Conservation Act (1096/1996) which allowed the CMPP to be implemented by means of land expropriations, including a full financial compensation or land exchange to landowners (Council of State 2012). The CMPP was later converted to a voluntary program, rejecting the option of expropriations (Salomaa et al. 2018; Nieminen et al. in review). However, before the rejection, landowners of mires with conservation potential were notified about the CMPP. The notification could have provoked owners of wooded mires to conduct logging in order to avoid their lands from being protected. Claims and anecdotes of such actions exist in social media sources like Twitter and forums of forestry magazines.

The aim of this paper is to determine if notifying landowners of their lands' conservation potential led to pre-emptive loggings on Finnish wooded mires. We analyzed whether harvesting rates of wooded mires chosen as candidate sites for the CMPP differed from harvesting rates of all other similar wooded mires in Finland that were not candidates for the CMPP. We also compared annual and monthly harvesting rates of mires with and without the candidate status to see whether events linked to the CMPP, such as notifying landowners of the conservation potential, caused sudden increase in the harvesting rates of the candidate wooded mires. To our knowledge, this is the first quantitative, nationwide analysis on pre-emptive behavior in Europe.

MATERIALS AND METHODS

Study case

In its preparation phase, the CMPP covered 327 300 ha of unprotected candidate mires considered for protection (Alanen and Aapala 2015; Kareksela et al. 2020). The aim was to protect about 100 000 ha of the ecologically most important mires to complement the existing mire protection network in Finland.

Originally in August 2012, the CMPP was politically agreed to be based on the Nature Conservation Act which enables land expropriations for conservation purposes (Council of State 2012). Practically, owners of the lands chosen for protection would have been allowed to decide whether to keep the ownership of the land, resulting in a private conservation area, or to sell it to the government. In both cases, landowners would have been compensated by being paid a market price for their land, or by exchanging their land for an equivalent parcel of the government's land elsewhere, depending on landowner's will.

The public briefing of the CMPP started in the beginning of 2013 by announcements in newspapers, a poll in a government-operated citizen portal in the internet, and hearings of stakeholder representatives. In May–July 2013, landowners of candidate mires received personal information letters notifying about field inventories that were made for the preparation of the CMPP during the summer 2013. In the autumn 2014, just before its implementation, the CMPP was revised to a voluntary program and the option of land expropriations was rejected due to political turmoil (Salomaa et al. 2018). This changed the CMPP's preparation and implementation remarkably. At that time, the CMPP provoked plenty of public deliberation. In the autumn 2015, 117 000 ha of the most ecologically valuable mires were proposed to be protected, but proper administrative tools to implement their protection did not exist.

Further political changes, such as cuts of conservation resources, left all but the government-owned proposed mires without protection. Afterwards, the CMPP has regularly appeared in the media and is mentioned also in the current Finnish Government Program (Anonymous 2019). In the current conservation policy, however, there are no signs of land expropriations being re-allowed in the CMPP.

Characteristics of wooded mires supporting their typical biodiversity features are connected to their tree stand and intact hydrological and microclimatic conditions (Laine et al. 1995; Maanavilja et al. 2014). Therefore, landowners resisting protection may easily impair conservation values of wooded mires with pre-emptive loggings. Landowners had, and still have, a possibility to log their wooded mires, since commercial forestry is legal on most of the mires considered to be included in the CMPP.

Study design

To ensure long-term effectiveness of conservation, candidate mires were planned to form hydrological entities (Aapala and Alanen 2015; Kareksela et al. 2020). Therefore, the candidate mires also included small patches of forests on mineral soils. We outlined the study to include only wooded habitat types occurring on peatlands in the boreal zone, i.e., spruce and pine mires. In Finland, both are commonly in a forestry use. For photographs of typical boreal spruce and pine mires, see Fig. S1.

We composed four groups of mires. The experimental group was composed of wooded mires with the candidate status and the control group of wooded mires without the status. Ideally, the study design would have included candidate mires of both informed and uninformed landowners, but in our real-world case all owners of candidate mires had been informed about their mires' conservation potential. Therefore, our study design was the best possible way to address the research questions.

Experimental and control groups were divided into spruce mires and pine mires. We analyzed harvesting rates during 5 years after the initial notification (2013–2017) and, additionally, stratified the data into annual and monthly harvesting rates. Concerning the monthly harvesting rates, we were especially interested in May–July 2013 when landowners received a notification of their lands being potential mires for protection, and October–November 2014 when the CMPP was revised to be a voluntary one. As a response variable for the overall harvesting rates over the 5 years and the annual harvesting rates, we calculated the area (hectares) that was logged and unlogged within each of the groups. As a response variable for the monthly harvesting rates, we utilized the number of submitted forest use notifications. We used the number rather than the hectares covered by the notifications because the sample sizes for monthly logged sites were small. In such a case the hectares might have masked the effect because an area covered by a notification varies, but a notification itself always reflects a landowner's behavior independent from the area. For figures showing the differences in the harvesting rates created according to logged area or submitted forest use notifications, see Appendix S1.

Candidate mires were mostly in a natural state or close to it. If they had been highly modified or degraded by human activity, they would not have been chosen as potential sites to the CMPP. Due to the desire to protect candidate mires as hydrological entities, some of them enclosed small degraded parts which were planned to be restored after protection. However, the average age and timber volume of candidate wooded mires likely represent those of older forests. To make the experimental and the control groups to be equivalent, we included to the analyses only those candidate and non-candidate wooded mires that were of the two most mature forest development class (advanced thinning stands and mature stands, see Appendix S2). We also calculated average diameters of trees in logged candidate and non-candidate mires and found that they did not differ remarkably, indicating that their timber quality was similar (Table S1).

We set the period of the study to be January 2013–December 2017. Since the CMPP was publicly briefed from January 2013 onwards, it was not reasonable to study harvesting rates earlier. If candidate wooded mires had been logged earlier, they would not have been selected as the candidates in the first place.

In Finland, the Forest Act (1093/1996) obliges forest owners to make a notification of forest use before logging. Practically, all loggings are executed after submitting a notification, since an industrial agent such as a timber buyer or a logging planner commonly makes the notification. Illegal loggings are very rare which is verified by a well-working law enforcement (Finnish Forest Centre 2018). Therefore, we applied notifications as surrogates for the loggings.

Data

For the analyses, we compiled eight different spatial data: unlogged and logged spruce and pine mires with the candidate status, and unlogged and logged spruce and pine mires without the status (Table 1).

To compile data of all wooded mires in Finland, we utilized publicly available spatial forest resource data which include detailed information of Finnish forests (https://urly.fi/1jgz). It covers the majority of privately owned forest land but mostly it does not include government- and municipalities-owned lands (Appendix S3). However, as the majority of forest land in Finland is

Table 1 Sample sizes of the final processed data

Candidate status	Habitat type	Logging status	Hectares	No. of notifications
Mires with the candidate status	Spruce mires	Unlogged	2198	na
		Logged	183	235
		Total	2381	235
	Pine mires	Unlogged	6661	na
		Logged	981	700
		Total	7642	700
Mires without the candidate status	Spruce mires	Unlogged	357 415	na
		Logged	78 196	54 314
		Total	435 611	54 314
	Pine mires	Unlogged	599 896	na
		Logged	136 390	61 473
		Total	736 286	61 473

privately owned (Official Statistics of Finland 2011–2016), the data coverage can be considered representative. To restrict the data of all wooded mires to cover only advanced thinning or mature spruce and pine mire stands, we outlined the forest resource data according to a habitat type and a forest development class.

To compile data of all logged wooded mires in Finland, we utilized publicly available spatial data of forest use notifications which include information of logged forest stands (https://urly.fi/1jgF) (Appendix S3). It served as the source for all the wooded mire stands that were advanced thinning or mature ones and logged in 2013–2017. However, many of the notifications lacked information of the habitat type since it is not an obligatory field in the notification. To complete the habitat type information, we joined the notification data with the abovementioned data of all wooded mires and set the latter to act as a primary source for the habitat type. However, the data of all wooded mires did not cover all stands in the forest use notification data. We checked whether the notifications on these stands included the habitat type information, and if they did, it was used as the source for the habitat type. If the habitat type information was not available in either of the data, we excluded the stand in question from the analysis.

We detached the data of all logged wooded mires from the data of all wooded mires after which we had four data: unlogged and logged spruce and pine mires covering whole Finland.

The final eight data of unlogged and logged non-candidate and candidate spruce and pine mires were compiled by detaching the candidate spruce and pine mires from the abovementioned data of all unlogged and logged wooded mires. We made this by means of a separate data that covered locations of the CMPP's candidate sites (Alanen and Aapala 2015; Kareksela et al. 2020).

Final data processing

Assembling the datasets caused multiple fragment stands that were too small to be real forest stands. We analyzed the size distributions of the forest stand fragments separately for all eight datasets and estimated that excluding stands ≤ 0.14 ha would reduce the number of artificial stands without eliminating many of the real small stands (Appendix S4). It is likely that we did not succeed in excluding all the artificial stands and likewise, we possibly excluded some of the existing small stands. However, we found no reason to expect any bias in the data caused by the exclusion and, therefore, consider the data to be reliable. All the data were processed with ArcMap 10.

Statistical analysis

We analyzed the harvesting rates per 5 years on mires with and without the candidate status and separately for pine and spruce mires against randomized harvesting rate distributions. To create the distributions, we set the total logged hectares of all pine and spruce mires to randomly locate on the whole area of the respective habitat. Randomization was performed with RStudio version 1.1.456 and replicated 1000 times. Replicates were compiled into a distribution describing how large proportion of logged hectares would randomly locate on the candidate mires of each habitat type. For the R-script, see Appendix S5.

RESULTS

7.7% (183 ha) of spruce mires and 12.8% (981 ha) of pine mires with the candidate status were logged based on hectares covered with submitted forest use notifications. Respective numbers for spruce mires without the candidate status were 18.0% (78 916 ha) and for pine mires 18.5% (136 390 ha). Therefore, the candidate mires were logged significantly less than the non-candidate ones (Fig. 1). For a map describing locations of all candidate mires and logged and unlogged wooded candidates, see Fig. S2.

Notifying landowners of their mires' conservation potential in May–July 2013 or revising the CMPP to a voluntary one in October–November 2014 did not produce harvesting peaks according to the annual harvesting rates that were calculated based on hectares covered with submitted forest use notifications (Fig. 2). In relation to the area of all candidate spruce or pine mires, the average annual harvesting rates on them were 1.54% and 2.57%, respectively. On candidate spruce mires, the harvesting rate was highest in 2013 (2.03% of all candidate spruce mires logged), whereas on pine mires, it was highest in 2016 (2.96% of all candidate pine mires logged). Candidate spruce mires were logged least in 2014 (0.96%) and pine mires in 2013 and 2017 (2.39% in both years). In relation to the area of all logged wooded mires, the logged area of candidate wooded mires was very low: on candidate spruce mires it varied between 0.14 and 0.32% and on candidate pine mires between 0.60 and 0.81%.

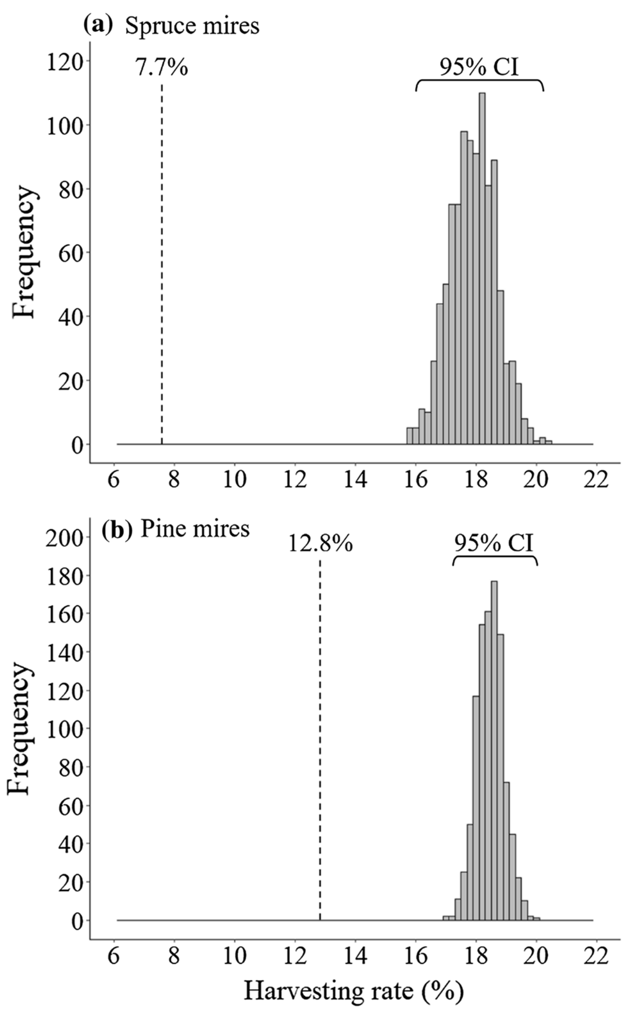


Fig. 1 Gray bars show how large proportion of logged hectares would randomly locate on candidate **a** spruce mires and **b** pine mires, when randomization is replicated 1000 times. Actual harvesting rates per 5 years on candidate mires are marked with dashed vertical lines

Within the years, both candidate and non-candidate mires had seasonal variation in numbers of submitted forest use notifications (Fig. 3). Notifications were submitted more in autumn and winter, and less in spring and summer. On candidate spruce mires, the highest numbers of notifications during the study period were submitted in October 2014 (11 notifications), in April 2013, and in January 2016 (9 notifications during both). Respective months and years for candidate pine mires were October 2017 (25 notifications), and October and November 2014 (24 notifications during both). Taking into account the seasonal variation, the numbers of submitted notifications did not peak in May–July 2013, when landowners were notified that their mires are candidates for the CMPP, nor in October–November 2014, when the option of land expropriations was rejected.

DISCUSSION

Our main finding was that notifying landowners of their mires' conservation potential and the possibility of mires becoming included in the CMPP did not cause systematic pre-emptive loggings. Instead, candidate wooded mires were logged significantly less than mires that were not considered for protection. The result is different from previous studies. In the USA, landowners have intentionally damaged species and habitats by applying shorter rotation times of loggings (Lueck and Michael 2003; Zhang 2004) and by changing land management practices (Brook et al. 2003). In Australia, pre-emptive behavior has caused loss of remnant forests (Simmons et al. 2018b).

While our results are encouraging, the root causes of the differences between our results and those of the earlier studies deserve further discussion. It is likely that landowners' behavior is shaped by the society they live in. Majority of Finns, regardless of their socioeconomic or demographic status, agree that protection of mire habitats and species is important (Tolvanen et al. 2013). This finding was supported also by a survey that was conducted to the citizen owners of the candidate mires during the preparation of the CMPP: almost half of the respondents had a positive attitude towards protection of their mires (Alanen and Aapala 2015). The rise of voluntary nature conservation following from Natura 2000 and other command-and-control approaches have likely helped to overcome previous biodiversity conflicts in Finland (Paloniemi and Vilja 2009), possibly making the public attitude receptive to new conservation initiatives. Furthermore, there is evidence that a fair conservation compensation decreases the likelihood of pre-emptive behavior and increases the likelihood of pro-conservation behavior (e.g., Langpap 2006; Ferraro et al. 2007; Byl 2019). According to Finnish legislation, landowners are eligible to a market price compensation of economic losses caused by land expropriations for biodiversity protection or for any other societally significant purposes. Hence, one reason for the lack of systematic pre-emptive behavior in our case may be that Finnish landowners perhaps do not feel their livelihood is seriously threatened by land use restrictions. In contrast, land use restrictions set e.g., by the Endangered Species Act in the USA are not compensated monetarily. Instead, if landowners pledge to certain conservation activities, they can be compensated e.g., by providing various assurances of not to set further restrictions on their land (Donahue 2005). Hence, principles of compensation are fundamentally different in Finland compared to the USA.

To determine if there were any obvious relationships between the CMPP's events and temporal patterns of the logging activity, we inspected yearly and monthly harvesting rates. The yearly harvesting rates were rather constant, but the monthly harvesting rates varied seasonally. This is explained by weather conditions favoring timber harvesting in autumn and winter when the ground is frozen and holds up forest harvesters. In May–July 2013, landowners of candidate wooded mires were notified that their mires are potential sites for protection. There was no detectable change in the monthly harvesting rates on candidate wooded mires relative to non-candidate ones, indicating that notifying landowners did not cause an increase in the logging activity. Another event potentially increasing the harvesting rates on candidate wooded mires was the decision of changing the CMPP to a voluntary program and rejecting the option of land expropriations in October–November 2014. After this, some landowners could have thought they had an opportunity to harvest without a disapproval of neighbors or the society at large (Jackson-Smith et al. 2005). To some landowners, the coverage of the conflict could have served as a reminder of the CMPP's preparation, or even as a support for defiance against biodiversity protection. However, there were again no obvious change in the monthly harvesting rates on candidate wooded mires relative to non-candidate ones.

In a comparative study like ours, there is always a possibility that some other factors than the ones being explored have had an impact on the response variable. In our case, the characteristics of tree stand on candidate wooded mires and their non-candidate counterparts were similar (Table S1), but candidate wooded mires had on average higher biodiversity values since they were chosen as potential protected areas. In order to host high biodiversity, candidate wooded mires could not have been largely exposed to former land use practices such as ditching or logging since mires' typical biodiversity features are dependent on tree stand and intact hydrology (Laine et al.

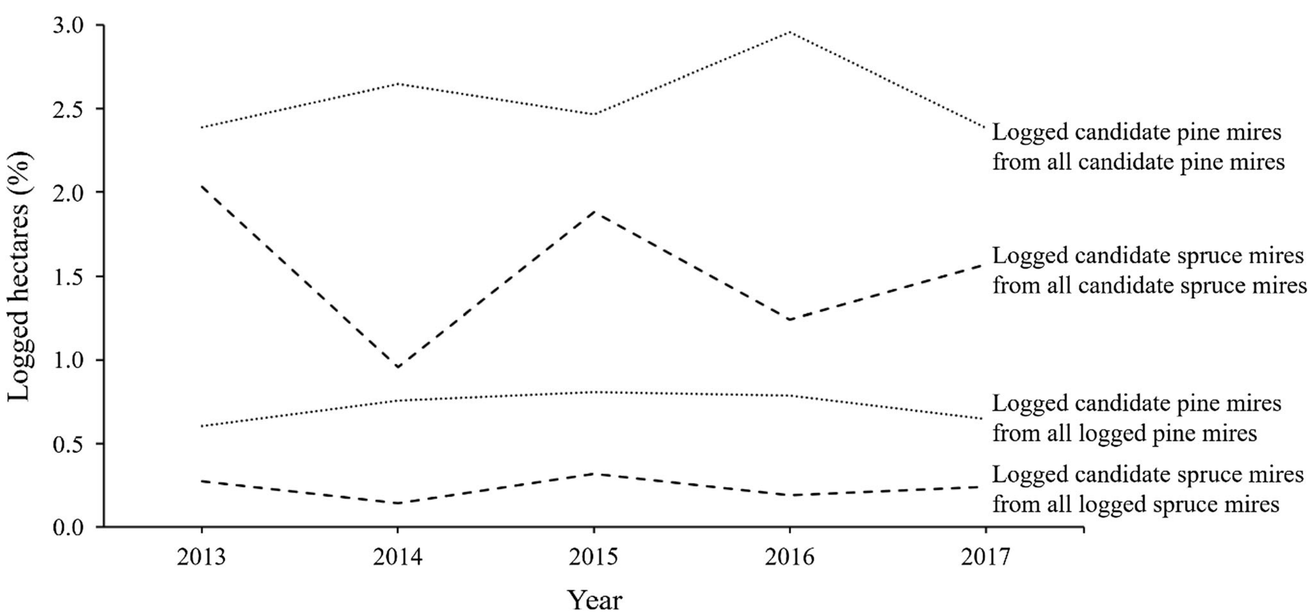


Fig. 2 Habitat-specific annual harvesting rates on candidate wooded mires calculated from all candidate wooded mires and from all logged wooded mires. Calculations of logged areas are based on hectares covered by submitted forest use notifications. Dashed lines represent spruce mires and dotted lines pine mires

1995; Maanavilja et al. 2014). At least two factors could explain why candidate wooded mires were less exposed to land use in the first place, possibly affecting also landowners' responses to informing about their mires' conservation potential. First, it is possible that candidate wooded mires could have been on average smaller sized than non-candidate ones. This is because in the era of heavy ditching campaign in the 1960s and 1970s (Vasander 2006), large mires having a high potential for wood production or peat mining were probably more likely ditched than smaller mires. Second, it might be that candidate wooded mires locate further away from roads than their non-candidate counterparts. If candidate wooded mires were on average smaller and/or more remote than non-candidate ones, they could have been silviculturally less attractive. In Finland, however, Forestry Management Associations often endeavor to centralize loggings to certain areas so that neighboring forest properties are logged at the same time, lowering the logistical costs of small-sized regeneration ready stands. This balances the effect of possibly smaller average size of candidate wooded mires on landowners' willingness to log. Unfortunately, we were not able to calculate the areas of single candidate or non-candidate mires since their borders were lost due to the data processing (see Materials and Methods). Furthermore, over 99% of forest land in central Finland locates < 400 m from the nearest road (Viitala et al. 2004) and the government supports construction of new forest roads in the whole country (Temporary Act on the Financing of Sustainable Forestry 34/2015), so it is improbable that remoteness would have prevented landowners to log their candidate wooded mires. Even if there were some other reasons for the candidate mires' low harvesting rates than landowners' awareness of their lands' conservation potential, the lack of obvious increases in the logging activity on candidate mires after notifying of their conservation value means that landowners did not engage in systematic pre-emptive loggings.

Despite the low harvesting rates, some of the candidate wooded mires were nevertheless logged in each study year. The rather constant yearly harvesting rates of candidate wooded mires, the seasonal variation in their harvesting rates imitating that of non-candidate wooded mires, and the lack of obvious logging peaks after notification letters implicate that instead of intentionally harming biodiversity values, there may have been some other reasons to log. Landowners may have simply followed their long-term logging plans that are often made in cooperation with local forestry specialists. Evidence shows that forestry-oriented landowners trust forestry specialists and prefer cooperating with them also in conservation issues rather than with environmental authorities (Paloniemi et al. 2006). Therefore, forestry-oriented landowners may have actively disregarded the information provided by the environmental authorities about their mires' high conservation potential. Such behavior may be expected particularly if a landowners' income is dependent on the actualized loggings. There is earlier evidence of Finnish forest owners

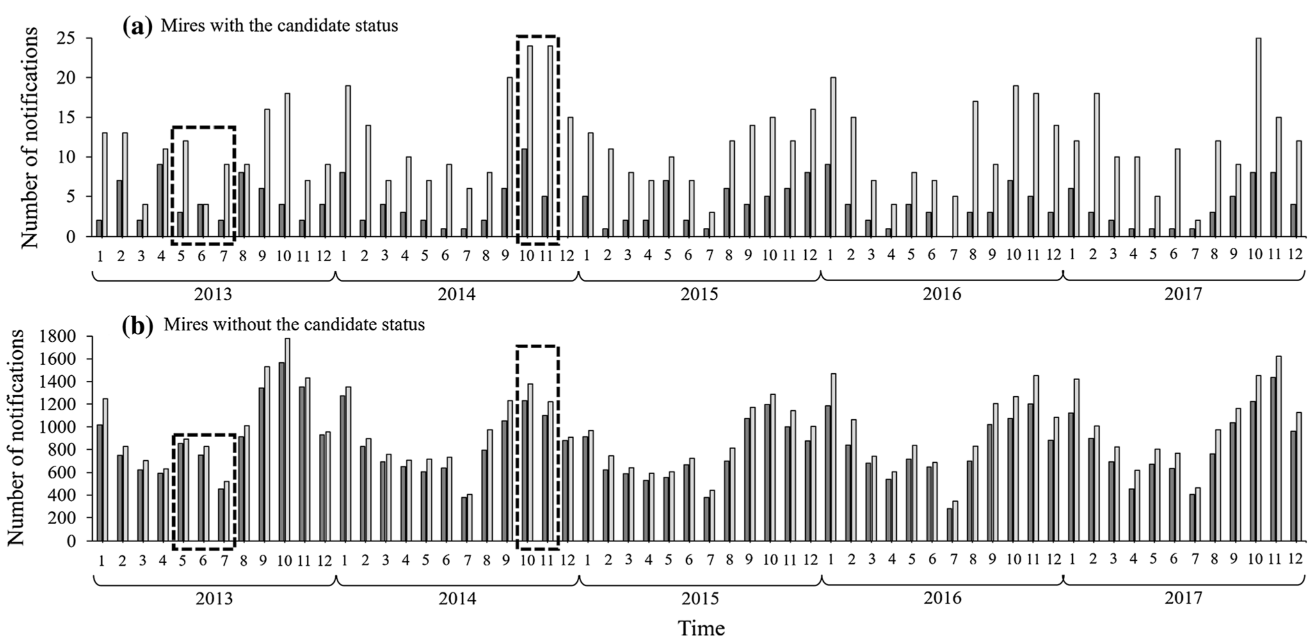


Fig. 3 Numbers of forest use notifications submitted per month in 2013–2017. Dark gray bars represent spruce mires and light gray bars pine mires. **a** Mires with the candidate status. **b** Mires without the candidate status. The boxes with dashed lines represent the months when we expected the number of notifications to rise due to the certain events concerning the CMPP

intentionally taking actions to harm flying squirrel (*Pteromys volans*) (Jokinen et al. 2018), so it is also possible that some owners of candidate wooded mires executed intentional pre-emptive loggings.

Since some mire owners are conservation minded (Alanen and Aapala 2015), it might be possible that at some point, logging candidate wooded mires would reduce even without protection. However, random factors such as transferring land property to the next generation or sudden acute need of money may initiate logging of a biodiversity-rich but non-protected mire even if the current owner would have decided to set the mire aside by his/her own decision. Excluding biodiversity-rich areas from official protection is a potential threat for long-term persistence of biodiversity, since forestry in Finland is so intensive that majority of forest sites will be logged when they reach maturity (Natural Resources Institute Finland 2019). If loggings on candidate wooded mires continued with the observed average annual rate of 36.6 ha (1.54%) for spruce mires and 1962 ha (2.57%) for pine mires, it would take only 26 and 13 years to loose half of them, respectively. Additionally, the likelihood to reach an ecologically representative mire conservation network decreases as increasingly larger area of candidate wooded mires are exposed to loggings. Originally, from 327 300 ha of candidate mires, 117 000 ha were proposed for protection (Alanen and Aapala 2015; Kareksela et al. 2020), leaving 210 300 ha without a protection request. Revising the CMPP to a voluntary program enabled landowners to refuse protection, inevitably changing the combination of mires applicable for protection (Nieminen et al. in review). In this new situation, 210 300 ha of mires originally not proposed to the CMPP could serve as compensatory sites for the mires that would be left out from protection due to some landowners' unwillingness to protect. Therefore, logging both candidate wooded mires included in and excluded from the most ecologically valuable ones is problematic.

CONCLUSIONS

Avoiding land use regulations by intentionally harming certain species or habitats has been proved to be a true phenomenon in the USA and Australia. We made the first quantitative exploration of pre-emptive behavior in Europe by studying logging behavior of landowners in Finland after they were notified that their wooded mires are candidate sites for a program that aims to extend the national mire conservation network. Unlike previous studies, we did not find evidence of systematic pre-emptive behavior. It is likely that landowners' responses to potential land use restrictions caused by biodiversity protection depend on the country- or region-specific administrative, political, and cultural circumstances such as previous experiences of biodiversity conservation or the compensation practices. It is also possible that silvicultural characteristics of wooded mires such as harvesting restricted mainly to periods of frozen ground, or on average lower value of peatland forests compared to mineral soil forests can affect landowners' behavior so that the results could have been different if the study was focused on mineral soils. Therefore, determining the exact reasons for the low harvesting rates of candidate wooded mires and the lack of systematic pre-emptive behavior demands further research such as mapping of landowners' attitudes, motives, and beliefs. Nevertheless, our results are encouraging in showing that informing landowners openly about their lands' conservation potential does not categorically lead to pre-empting of conservation values on wooded mires.

Acknowledgements The authors thank Dr. Anssi Lensu for counseling on data and analyses. Eini Nieminen was funded by Maj and Tor Nessling Foundation and Kone Foundation. Kalle Salovaara was funded by The Finnish Foundation for Nature Conservation. Open access funding provided by University of Jyväskylä (JYU).

REFERENCES

Alanen, A., and K. Aapala. 2015. Proposal of the Mire Conservation Group for supplemental mire conservation. Reports of the Ministry of the Environment 26/2015 (in Finnish, English summary). http://hdl.handle.net/10138/158285.

Alphandéry, P., and A. Fortier. 2001. Can a territorial policy be based on science alone? The system for creating the Natura 2000 network in France. *Sociologia Ruralis* 41: 311–328. https://doi.org/10.1111/1467-9523.00185.

Anonymous. 2019. Inclusive and competent Finland—a socially, economically and ecologically sustainable society. Programme of Prime Minister Sanna Marin's Government 2019. Retrieved April 20, 2020, from https://valtioneuvosto.fi/en/rinne/government-programme.

Blicharska, M., E.H. Orlikowska, J.M. Roberge, and M. Grodzinska-Jurczak. 2016. Contribution of social science to large scale biodiversity conservation: A review of research about the Natura

2000 network. *Biological Conservation* 199: 110–122. https://doi.org/10.1016/j.biocon.2016.05.007.
Brook, A., M. Zint, and R. De Young. 2003. Landowners' responses to an endangered species act listing and implications for encouraging conservation. *Conservation Biology* 17: 1638–1649. https://doi.org/10.1111/j.1523-1739.2003.00258.x.
Byl, J.P. 2019. Perverse incentives and safe harbors in the endangered species act: Evidence from timber harvests near woodpeckers. *Ecological Economics* 157: 100–108. https://doi.org/10.1016/j.ecolecon.2018.11.008.
Council of State. 2012. The Government Resolution on the Sustainable Use and Protection of Peatlands. Retrieved January 15, 2012, from https://urly.fi/1mu5 (in Finnish).
Council of State. 2014. Decision in principle about continuing the program protecting biodiversity of forests in South Finland. Retrieved April 20, 2014, from https://urly.fi/1Anj (in Finnish).
Donahue, D. 2005. The Endangered Species Act and its current set of incentive tools for species protection. In *Species at risk: Using economic incentives to shelter endangered species on private lands*, ed. J.F. Shogren, 25–63. Austin: University of Texas Press.
Ferraro, P.J., C. Mcintosh, and M. Ospina. 2007. The effectiveness of the US endangered species act: An econometric analysis using matching methods. *Journal of Environmental Economics and Management* 54: 245–261. https://doi.org/10.1016/j.jeem.2007.01.002.
Finnish Forest Centre. 2018. Law enforcement of Forest Acts in 2018. Retrieved April 20, 2018, from https://www.metsakeskus.fi/sites/default/files/lainvalvonta-2018.pdf (in Finnish).
Gibbs, K.E., and D.J. Currie. 2012. Protecting endangered species: Do the main legislative tools work? *PLoS ONE* 7: e35730. https://doi.org/10.1371/journal.pone.0035730.
Grodzinska-Jurczak, M., and J. Cent. 2011. Expansion of nature conservation areas: Problems with natura 2000 implementation in Poland? *Environmental Management* 47: 11–27. https://doi.org/10.1007/s00267-010-9583-2.
Hiedanpää, J. 2002. European-wide conservation versus local well-being: The reception of the Natura 2000 Reserve Network in Karvia, SW-Finland. *Landscape and Urban Planning* 61: 113–123. https://doi.org/10.1016/S0169-2046(02)00106-8.
IPBES. 2019. Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. In ed. S. Díaz, J. Settele, E.S. Brondízio, H.T. Ngo, M. Guèze, J. Agard, A. Arneth, P. Balvanera, et al. IPBES secretariat: Bonn.
Jackson-Smith, D., U. Kreuter, and R.S. Krannich. 2005. Understanding the multidimensionality of property rights orientations: Evidence from Utah and Texas Ranchers. *Society and Natural Resources* 18: 587–610. https://doi.org/10.1080/08941920590959578.
Jokinen, M., T. Hujala, R. Paloniemi, and A. Vainio. 2018. Private landowners and protected species: What sort of noncompliance should we be worried about? *Global Ecology and Conservation* 15: e00407. https://doi.org/10.1016/j.gecco.2018.e00407.
Kabii, T., and P. Horwitz. 2006. A review of landholder motivations and determinants for participation in conservation covenanting programmes. *Environmental Conservation* 33: 11–20. https://doi.org/10.1017/S0376892906002761.
Kamal, S., M. Kocór, and M. Grodzińska-Jurczak. 2015. Conservation opportunity in biodiversity conservation on regulated private lands: Factors influencing landowners' attitude. *Environmental Science & Policy* 54: 287–296. https://doi.org/10.1016/j.envsci.2015.07.023.
Kareksela, S., K. Aapala, A. Alanen, T. Haapalehto, J.S. Kotiaho, J. Lehtomäki, N. Leikola, N. Mikkonen, A. Moilanen, E. Nieminen, S. Tuominen, and R. Virkkala. 2020. Combining spatial prioritization and expert knowledge facilitates effectiveness of large-scale mire protection process in Finland. *Biological Conservation*. https://doi.org/10.1016/j.biocon.2019.108324.
Keulartz, J. 2009. European nature conservation and restoration policy—Problems and perspectives. *Restoration Ecology* 17: 446–450. https://doi.org/10.1111/j.1526-100X.2009.00566.x.
Knight, R.L. 1999. Private lands: The neglected geography. *Conservation Biology* 13: 223–224. https://doi.org/10.1046/j.1523-1739.1999.013002223.x.
Laine, J., H. Vasander, and R. Laiho. 1995. Long-term effects of water level drawdown on the vegetation of drained pine mires in southern finland. *Journal of Applied Ecology* 32: 785–802.
Lambin, E.F., and P. Meyfroidt. 2011. Global land use change, economic globalization, and the looming land scarcity. *Proceedings of the National Academy of Sciences* 108: 3465–3472. https://doi.org/10.1073/pnas.1100480108.
Langpap, C. 2006. Conservation of endangered species: Can incentives work for private landowners? *Ecological Economics* 57: 558–572. https://doi.org/10.1016/j.ecolecon.2005.05.007.
Lueck, D., and J.A. Michael. 2003. Preemptive habitat destruction under the Endangered Species Act. *The Journal of Law and Economics* 46: 27–60. https://doi.org/10.1086/344670.
Maanavilja, L., K. Aapala, T. Haapalehto, J.S. Kotiaho, and E. Tuittila. 2014. Impact of drainage and hydrological restoration on vegetation structure in boreal spruce swamp forests. *Forest Ecology and Management* 330: 115–125. https://doi.org/10.1016/j.foreco.2014.07.004.
Natural Resources Institute Finland. 2019. Statistics database. Age of forest stands on forest land. Retrieved November 12, 2019, from https://urly.fi/1mt3.
Newbold, T., L.N. Hudson, S.L.L. Hill, S. Contu, I. Lysenko, R.A. Senior, L. Börger, D.J. Bennett, et al. 2015. Global effects of land use on local terrestrial biodiversity. *Nature* 520: 45–50. https://doi.org/10.1038/nature14324.
Norton, D.A. 2000. Conservation biology and private land: Shifting the focus. *Conservation Biology* 14: 1221–1223.
Official Statistics of Finland 2011–2016. Ownership of Forest Land. e-publication. Helsinki: Natural Resources Institute Finland. Retrieved November 12, from https://stat.luke.fi/en/ownership-forest-land.
Olive, A. 2016. It is just not fair: The Endangered Species Act in the United States and Ontario. *Ecology and Society* 21: 13. https://doi.org/10.5751/ES-08627-210313.
Paavola, J. 2003. Protected areas governance and justice: Theory and the European Union's Habitats Directive. *Environmental Sciences* 1: 59–77. https://doi.org/10.1076/evms.1.1.59.23763.
Paloniemi, R., I. Massa, and P. Tikka. 2006. Forest owners and official nature protection. Maaseudun uusi aika 3/2006. Retrieved April 20, 2006, from https://urly.fi/1muh (in Finnish).
Paloniemi, R., and V. Vilja. 2009. Changing ecological and cultural states and preferences of nature conservation policy: The case of nature values trade in South-Western Finland. *Journal of Rural Studies* 25: 87–97. https://doi.org/10.1016/j.jrurstud.2008.06.004.
Pereira, H.M., P.W. Leadley, V. Proença, R. Alkemade, J.P.W. Scharlemann, J.F. Fernandez-Manjarréz, M.B. Araújo, P. Balvanera, et al. 2010. Scenarios for global biodiversity in the 21st century. *Science* 330: 1496–1501. https://doi.org/10.1126/science.1196624.
Salomaa, A., R. Paloniemi, and A. Ekroos. 2018. The case of conflicting Finnish peatland management—Skewed representation of nature, participation and policy instruments. *Journal of Environmental Management* 223: 694–702. https://doi.org/10.1016/j.jenvman.2018.06.048.

Schwartz, M.W. 2008. The performance of the Endangered Species Act. *Annual Review of Ecology Evolution and Systematics* 39: 279–299. https://doi.org/10.1146/annurev.ecolsys.39.110707.173538.

Silvasti, T. 2003. The cultural model of "the good farmer" and the environmental question in Finland. *Agriculture and Human Values* 20: 143–150. https://doi.org/10.1023/A:1024021811419.

Simmons, B.A., E.A. Law, R. Marcos-Martinez, B.A. Bryan, C. McAlpine, and K.A. Wilson. 2018a. Spatial and temporal patterns of land clearing during policy change. *Land Use Policy* 75: 399–410. https://doi.org/10.1016/j.landusepol.2018.03.049.

Simmons, B.A., R. Marcos-martinez, E.A. Law, B.A. Bryan, and K.A. Wilson. 2018b. Frequent policy uncertainty can negate the benefits of forest conservation policy. *Environmental Science & Policy* 89: 401–411. https://doi.org/10.1016/j.envsci.2018.09.011.

Tolvanen, A., A. Juutinen, and R. Svento. 2013. Preferences of local people for the use of peatlands: The case of the richest peatland Region in Finland. *Ecology and Society* 18: 19. https://doi.org/10.5751/ES-05496-180219.

Vasander, H. 2006. The use of mires for agriculture and forestry. In *Finland—land of mires*, ed. T. Lindholm, and R. Heikkilä, 173–178. The Finnish Environment Institute: Helsinki. http://hdl.handle.net/10138/37961.

Viitala, E.-J., V.-M. Saarinen, A. Mikkola, and M. Strandström. 2004. Determining construction need for forest roads by means of spatial data. Metsätieteen aikakauskirja 2/2004: 175–192 (in Finnish). http://urn.fi/URN:NBN:fi-fe2016111628790.

Zhang, D. 2004. Endangered species and Timber Harvesting—The case of red-cockaded woodpeckers. *Economic Inquiry* 42: 150–165. https://doi.org/10.1093/ei/cbh051.

AUTHOR BIOGRAPHIES

Eini Nieminen (✉) graduated as a Doctor of Philosophy at the University of Jyväskylä, Finland. Her research interests include spatial conservation planning and reconciling the needs of nature and human societies.
Address: Department of Biological and Environmental Science, University of Jyväskylä, P.O. Box 35, 40014 Jyväskylä, Finland.
Address: School of Resource Wisdom, University of Jyväskylä, P.O. Box 35, 40014 Jyväskylä, Finland.
e-mail: eini.m.nieminen@jyu.fi

Kalle Salovaara graduated as a Master of Science at the University of Jyväskylä, Finland. His research interests include sustainable land use and forest management.
Address: Department of Biological and Environmental Science, University of Jyväskylä, P.O. Box 35, 40014 Jyväskylä, Finland.
e-mail: kallesalovaara94@gmail.com

Panu Halme is Senior Lecturer at the University of Jyväskylä, Finland. His research interests include the conservation ecology of several species groups in forested ecosystems, traditional rural biotopes, peatlands, and wetlands.
Address: Department of Biological and Environmental Science, University of Jyväskylä, P.O. Box 35, 40014 Jyväskylä, Finland.
Address: School of Resource Wisdom, University of Jyväskylä, P.O. Box 35, 40014 Jyväskylä, Finland.
e-mail: panu.halme@jyu.fi

Janne Sakari Kotiaho is Professor of Ecology at University of Jyväskylä, Finland. His research interests include the intersection of ecology, conservation biology and societal decision making.
Address: Department of Biological and Environmental Science, University of Jyväskylä, P.O. Box 35, 40014 Jyväskylä, Finland.
Address: School of Resource Wisdom, University of Jyväskylä, P.O. Box 35, 40014 Jyväskylä, Finland.
e-mail: janne.kotiaho@jyu.fi

III

QUANTIFYING TRADE-OFFS BETWEEN ECOLOGICAL GAINS, ECONOMIC COSTS, AND LANDOWNERS' PREFERENCES IN BOREAL PEATLAND PROTECTION

by

Eini Nieminen, Santtu Kareksela, Panu Halme & Janne Kotiaho 2020

Submitted manuscript

Request a copy from author.